



Accumulation and fate of microplastics in soils after application of biosolids on land: A review

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Abstract

The recent discovery of microplastic occurrence in most ecosystems is raising health concerns globally, yet the fate of microplastics is poorly known, particularly in the terrestrial environment. Here, we review the effect of land applications of biosolids on microplastic contamination in agricultural soils. We focus on separation and identification methods, distribution in wastewater, and retention, migration and degradation in soils. We also discuss the uptake of microplastics by microorganisms. The number of microplastic particles in biosolids ranges from 506 to 15,385 per kg of biosolids. After biosolid application on land, microplastic number ranges from 18 to 6.9×10^5 particles per kg of soil, depending on the soil composition. Microplastic retention in soils increases with soil organic carbon concentration, Fe and Al oxide concentrations, and soil ionic strength. Biodegradation of microplastics causes mass loss, changes in chemical composition, reduction in molecular weight and mechanical strength, and gas production of carbon dioxide and methane.

Keywords Microplastic · Soil · Biosolids · Migration · Degradation · Biological uptake

Introduction

Plastic is a synthetic material that has been widely used in packaging, transportation, electronics, building, and construction, due to its durability, low production cost, and lightweight, with an annual global plastic production of 359 million metric ton in 2018 (Geyer et al. 2017; Ho et al. 2018; Okoffo et al. 2021), and it is expected to increase to 1,200 million metric ton by 2050 (Dai et al. 2022b). Microplastics are defined as plastic fragments or particles ranging in size from 100 nm to 5 mm, whereas nanoplastics are defined as plastic fragments or particles smaller than 100 nm (Nguyen et al. 2019; Boyle and Örmeci 2020; Sharma et al. 2022). The major sources of microplastics are the fragmentation of large plastics, personal care, cosmetic products, and other industrial applications (Li et al. 2018; Guo et al. 2020; Atugoda et al. 2022; Mohamed et al. 2022a). Microplastics have

been detected in stormwater, household sewage, and land-fill leachate that enter the influent stream to be treated at wastewater treatment plants before entering receiving water sources (Mahon et al. 2017; Padervand et al. 2020; Dai et al. 2022a; Stang et al. 2022). The global microplastic concentrations in wastewater influent range from 0.28 to 3.14×10^4 particles per litre (Liu et al. 2021).

Wastewater treatment plants have different microplastic removal efficiencies because of the implementation of different combinations of treatment processes in the primary, secondary, and tertiary stages, such as filtration, sedimentation, flotation, and coagulation (Raju et al. 2018; Padervand et al. 2020; Hou et al. 2021). For example, the coagulation process alone can remove 47.1 to 81.6% of microplastics from wastewater influent (Hou et al. 2021; Khan et al. 2022). Overall, up to 99% of microplastics are removed from wastewater influents at wastewater treatment plants and retained in sewage sludge (Okoffo et al. 2021). Biosolids are produced after the thickening (Mohamed et al. 2023), digestion, and dewatering of sewage sludge to destroy pathogens and stabilise contaminants (Mohamed and Li 2022; Mohamed et al. 2022b). However, the produced biosolids can contain approximately 80% microplastics from untreated sludge, with 20% microplastics remaining in the effluent, continuously entering the ecosystem where they can bioaccumulate

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(Raju et al. 2018; Hou et al. 2021). Biosolids are nutrient-rich waste products that are widely used as soil amendment agents or fertilisers in agricultural lands in many countries (Nizzetto et al. 2016; Okoffo et al. 2020a; Zhuang et al. 2022). High percentages of biosolids have been used for agricultural applications in Norway (82%), Australia (75%), Ireland (63%), and the USA (55%) (Figure S1, Supplementary Material) (Liu et al. 2021). Besides, in Finland, 89% of the biosolids are used for composting; and in the Netherlands, 99% of the biosolids are incinerated (Rolsky et al. 2020). China and Finland both have about 35% of the biosolids going towards landfills (Rolsky et al. 2020). Biosolids contain most microplastics from wastewater, which could be a major source of accumulated microplastics in agricultural soils.

Consequently, an increasing number of studies have reported an increase in microplastic accumulation in soil environments after biosolids or sewage sludge application in agriculture (Crini and Lichtfouse 2019; Rolsky et al. 2020; Chia et al. 2021; Morin-Crini et al. 2022). Since microplastics enter agricultural soils through biosolids or sludge application, they can migrate vertically into deeper soil layers and contaminate subsurface receptors, such as groundwater and other underground ecosystems (O'Connor et al. 2019; Crossman et al. 2020; Guo et al. 2020; Mohajerani and Karabatak 2020). During dry seasons, wind can disperse microplastics widely in the soil environments (Gatidou et al. 2019; Padervand et al. 2020; Yang et al. 2021).

Studies have shown that microplastics cause bioaccumulation in the food webs (Mahon et al. 2017; Raju et al. 2018; Alava 2020). For instance, Shabaka et al. recovered 7,000 microplastic particles in fish from the Mediterranean coast of Egypt during their investigation (Shabaka et al. 2020). A recent study analysed six human placentas to evaluate the presence of microplastics; 12 microplastic fragments were found in four human placentas (Ragusa et al. 2021). Additionally, microplastics can remain within the digestive tract or migrate into the bloodstream of animals, along with other contaminants and chemicals, such as bisphenol A and phthalates attached to plastic particles, either during production or disposal. Biomagnification occurs as species from higher trophic levels consume other species containing microplastics, eventually resulting in a public health risk when people consume foods contaminated with microplastics (Clarke and Smith 2011; Miller et al. 2020). As microplastics have become a major concern because of their global health risks, it is necessary to study the increasing microplastic concentration in soils to understand the fate of microplastics in different soil environments and identify and address the research gaps in future studies.

Because these studies focused on the effects of different soil characteristics on microplastic migration processes, understanding the efficiency of microplastic degradation

in soil environments is also extremely limited. Thus, more studies on this topic would contribute to the understanding of how microplastics bioaccumulate in aquatic organisms and soil biota. To the best of our knowledge, no previous review articles have studied and analysed microplastic concentrations and polymer types in biosolids and soil samples. Furthermore, the migration and biodegradation processes of microplastics in agricultural soils have not been previously discussed. We hypothesised that the types and concentrations of plastic polymers in biosolids could also directly affect how microplastics enter soil environments, which consecutively affects bioaccumulation and contamination of microplastics in the ecosystem. Moreover, we hypothesised that the polymer distributions in wastewater influent and effluent could indicate microplastic polymer types in biosolids, which is also important for review. Compared with other literature reviews, this review describes microplastic accumulation, migration, and degradation processes and summarises the primary results from the current research to discuss different factors that may influence the fate of microplastics in soil environments. This review also discusses the extent of microplastic pollution due to biosolid applications on land.

Therefore, this review article discusses the vertical migration and biodegradation processes of microplastics in soils and other potential degradation processes, such as photodegradation and thermal degradation. The objectives of this review are to: (1) discuss the reported studies on the microplastic accumulation in the soil after biosolid applications, (2) compare the microplastic concentrations and polymer types in different regions and countries, (3) summarise the results of microplastic migration and degradation processes in soils, and (4) discuss the factors affecting the retention and migration of microplastics in soil environments. Microplastic concentrations and polymer distributions in different countries were compared to determine how local industries, population densities, and if a country is developed may affect polymer distribution. Discussing different microplastic removal efficiencies for wastewater treatment plants in different countries is beyond the scope of this review. This study will also help raise awareness about potential environmental threats resulting from microplastic pollution through biosolid land applications, which will help establish new regulations for biosolid applications in agricultural soils to prevent adverse toxicological impacts.

Separation and identification techniques

Most studies followed similar procedures to extract and identify microplastics in sludge and soil samples. The steps include sample collection and organic matter digestion using an oxidation reagent, such as H_2O_2 or Fenton's reagent, to

remove organic matter (Lares et al. 2018; Liu et al. 2018; Hurley et al. 2018b; Li et al. 2019; Long et al. 2019; Crossman et al. 2020). The particles are then separated via density separation by adding a salt-saturated solution or in combination with the elutriation technique (Mahon et al. 2017; Corradini et al. 2019; Li et al. 2019; Long et al. 2019; Rolsky et al. 2020). Sample identification is commonly conducted using stereomicroscope analysis, such as Fourier transform infrared spectroscopy, equipped with an attenuated total reflectance (Mintenig et al. 2017; Li et al. 2018, 2019; Crossman et al. 2020; Rolsky et al. 2020; Huang et al. 2022).

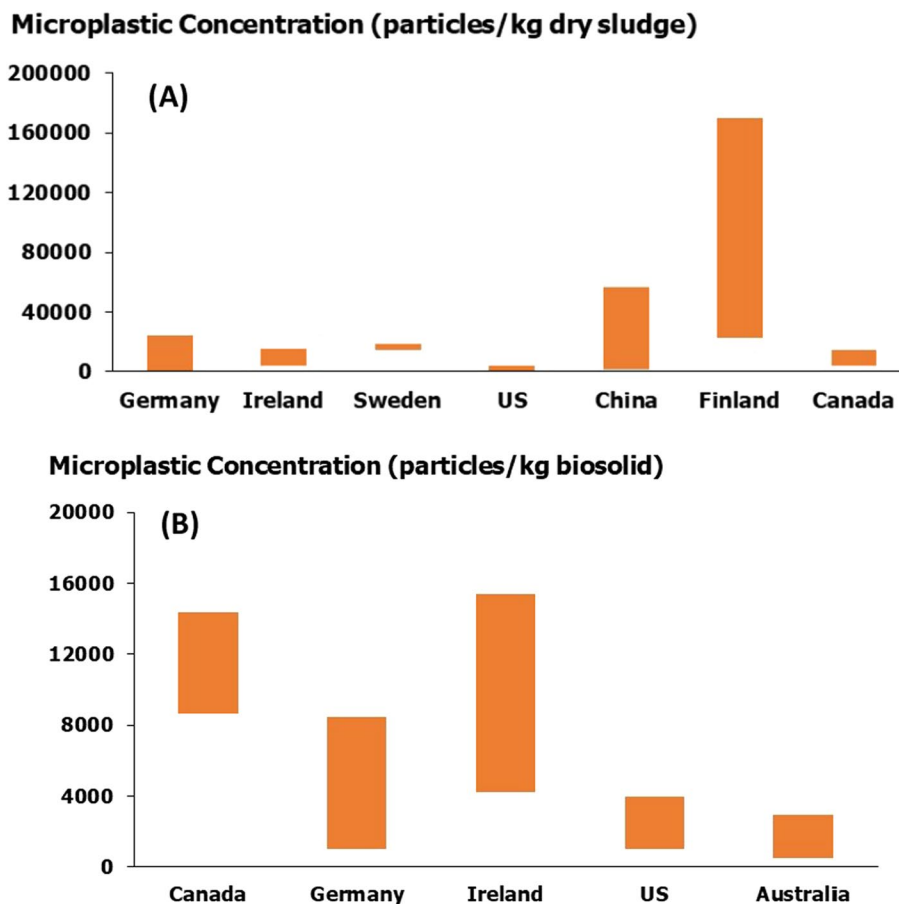
Some studies have reported using pressurised liquid extraction techniques for microplastic separation (Fuller and Gautam 2016; Dierkes et al. 2019; Okoffo et al. 2020a). Studies focused on sampling, separation, and identification methods for microplastics have been critically analysed. However, there are other variations among different studies, including the studied size ranges of microplastics and the characteristics of the examined soil samples in different regions and countries, which make comparisons more difficult. It should be noted that some of the microplastic concentrations detected in soils and reported in the literature may have entered soil environments through sources other

than biosolids, including agricultural compost, plastic film mulching, or other industrial activities.

Microplastic abundance and polymer distributions in biosolids

Figure 1A presents the range of microplastic concentrations in biosolid samples from different countries; the results were based on the different size ranges of microplastic particles reported in various studies (Zubris and Richards 2005; Magnusson and Norén 2014; Lassen et al. 2015; Carr et al. 2016; Mahon et al. 2017; Gies et al. 2018; Lares et al. 2018; Li et al. 2018). As shown, the samples collected in Finland possessed a wide range of microplastic concentrations, with a distinctly high maximum value of 170,000 particles per kg of dry sewage sludge, followed by China, Germany, Ireland, and Canada (Lares et al. 2018). After being produced at wastewater treatment plants, sludge containing microplastics requires thickening, digestion, and dewatering processes to destroy pathogens and stabilise contaminants to meet the quality standards for land applications (Hou et al. 2021). In the USA, biosolid application is regulated by Title 40, Code

Fig. 1 Range of microplastic concentrations in **A** sludge and **B** biosolid samples in different countries. Data sources for Fig. 1A: (Zubris and Richards 2005; Magnusson and Norén 2014; Lassen et al. 2015; Carr et al. 2016; Mahon et al. 2017; Gies et al. 2018; Lares et al. 2018; Li et al. 2018), and Fig. 1B: (Mahon et al. 2017; Crossman et al. 2020; Mohajeri and Karabatak 2020). Finland has very high microplastic concentrations in dry sludge from wastewater treatment plants with a wide range of concentrations. Canada and Ireland both possess extremely high microplastic concentrations in biosolids produced from wastewater treatment plants



of Federal Regulations Part 503 Biosolids Rule, in which biosolid quality requirements are established, but the maximum biosolid application rates are not well defined (Lu et al. 2012). Thus, the quantity of biosolids applied to agricultural soils should be regulated to control the accumulation of microplastics in soils.

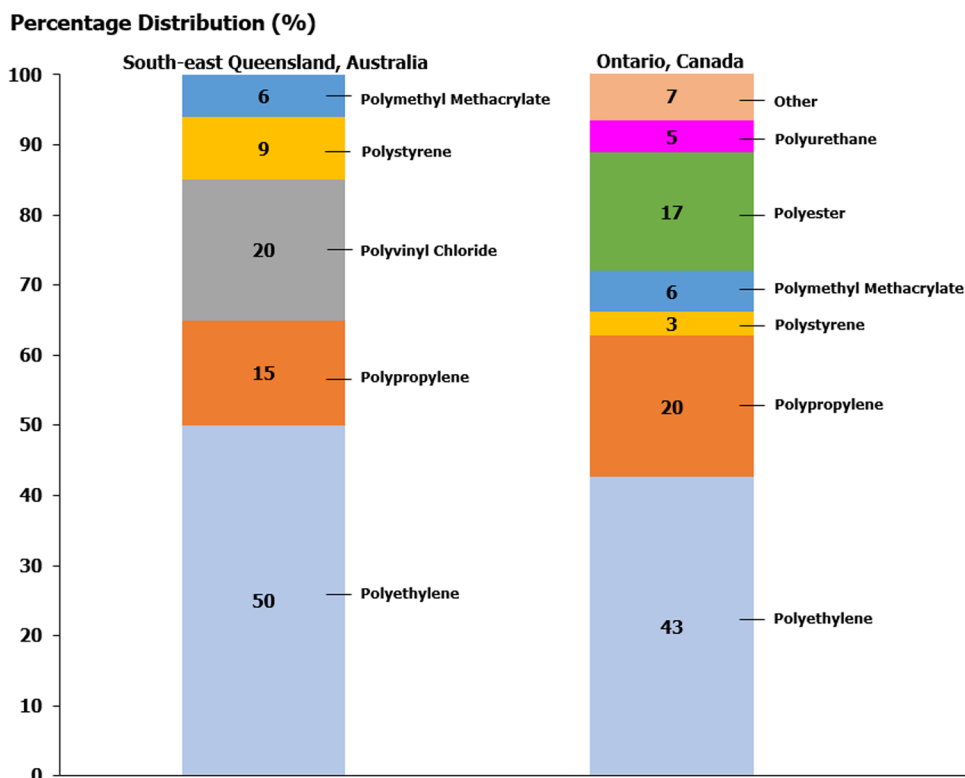
Many recent studies have shown that biosolids are becoming a significant source of microplastics in agricultural soils (Crossman et al. 2020; Rolsky et al. 2020; Okoffo et al. 2021; Rios Mendoza et al. 2021). Figure 1B presents the different microplastic concentrations in biosolid samples collected from different countries (Mahon et al. 2017; Crossman et al. 2020; Mohajerani and Karabatak 2020). A calculation can be performed to illustrate the significance of the amount of sludge-based microplastics applied to soils, considering biosolid production per year. For example, in Canada, with approximately 660,000 tons of annual biosolids production, an average microplastic concentration of 11.5 million particles per ton of biosolids, and a biosolids application rate of 47% on agricultural lands, a total amount of 3.6×10^{12} microplastic particles would be introduced into soil environments (Canadian Council of Ministers of the Environment 2012; Liu et al. 2018; Crossman et al. 2020). With an average mass of 2.45 mg per microplastic particle (Mohajerani and Karabatak 2020) and based on the previously estimated values, a total of 8,740 tons of microplastics are estimated to contaminate agricultural lands in Canada every year through biosolids application. However, no distinct relationship was

identified between the population density, gross domestic product, and microplastic concentrations in sludge or biosolids.

Figure 2 shows the average of microplastic polymer distributions in biosolid samples from Australia and Canada. Polyethylene and polypropylene polymers were abundant in both distributions. However, the compositions of the remaining microplastics differed. The biosolid samples collected in Australia contained more polyvinyl chloride, whereas those from Canada contained more polyester. The differences in polymer distribution may be caused by local industries. For example, samples collected from wastewater treatment plants in Canada received industrial wastewater inputs from the automotive industry, manufacturers of mining and railway equipment, and other plastic producers (Crossman et al. 2020).

Polypropylene and polyurethane polymers are commonly used in light vehicle production. Polyesters are commonly used to make automotive interiors, which may explain their abundance in biosolid samples (Maddah 2016; Economics & Statistics Department 2019; Kausar 2019). Okoffo et al. identified an abundance of polyethylene in biosolid samples collected in Queensland, Australia, which is most likely due to the use of personal care and cosmetic products, such as toothpaste and facial cleansers (Okoffo et al. 2020b). However, the high polyvinyl chloride content may be due to the use of construction materials, as polyvinyl chloride is commonly used to produce pipes, floor coverings,

Fig. 2 The average of microplastic polymer distribution in biosolid samples in Southeast Queensland, Australia (Okoffo et al. 2020b) and Ontario, Canada (Crossman et al. 2020). Polyethylene and polypropylene are abundant in Australia and Canada. The prevalence of polyethylene polymer could be attributed to the widespread use of personal care and cosmetic products



cable insulation, and roofing sheets (Mukesh and Anil 2005; Okoffo et al. 2020b). Overall, the types and distributions of microplastic particles vary widely among countries.

Microplastic accumulation and polymer types in soils

Once biosolids are applied to agricultural lands, microplastics enter the soil simultaneously. Table 1 summarises microplastic concentrations in soils with previous applications of biosolids from different studies. Corradini et al. reported that microplastic concentrations in soil increased with successive biosolid applications (Corradini et al. 2019). Similarly, Crossman et al. found higher residual microplastic levels at sites with two previous land applications in 2013 and 2015 than at sites without or with fewer prior occasions (Crossman et al. 2020). These findings agree with other studies in the literature, in which microplastics accumulated in soil environments after biosolid applications on land (Keller et al. 2020; van den Berg et al. 2020; Okoffo et al. 2021; Yang et al. 2021), potentially indicating that higher biosolid application rates were implemented than the actual migration and degradation rates of microplastics in soils. Van den Berg et al. found that soils with up to eight prior biosolids applications may average ~256% higher microplastic concentrations than soils without biosolids applications, with roughly a 32% increase in microplastics after each biosolid application (van den Berg et al. 2020). This agrees with the value reported by Corradini et al., in which there was a 34.8% increase in microplastic concentration in soil samples following each biosolid application (Corradini et al. 2019).

All reviewed studies reported microplastic accumulation in soils after biosolids application, except for one study by Crossman et al., who collected over 1300 soil samples from three different sites (Crossman et al. 2020).

It was found that the average net change in microplastic concentrations in soil at one site was positive (41%), while the microplastic concentrations for the other two sites decreased by 30 and 45%, respectively (Crossman et al. 2020). The negative net change in microplastic accumulation may be caused by different soil compositions, such as clay, sand, marsh, biosolids, or other external factors, such as high precipitation during the study period (Zemke et al. 2019; Crossman et al. 2020). Only one recent study directly showed that microplastic abundance in the soil is positively correlated with the population density of a region but negatively correlated with land elevation (Zhou et al. 2021). The effect of population density on the microplastic abundance in soils agrees with observations reported in previous studies (Lu et al. 2012; Scheurer and Bigalke 2018). However, other factors could affect this positive correlation with population density, such as sampling sites, local industries, weather conditions, rainfall intensity, and using stormwater treatment systems, which should be considered.

The relationship between microplastic abundance and land elevation may be explained by the vertical migration of microplastics from high-elevation areas to flatter ground (Zhou et al. 2021). This may also be explained by the preferred locations of agricultural lands at lower elevations compared with mountainous areas. Therefore, more biosolids were applied to plain soils (Haddaway et al. 2014). Easily accessible land might lead to more applications of biosolids in agricultural lands. However, we believe that other sources of microplastics, such as mulching, landfills, and other agricultural activities, can also contribute to microplastic abundance in soils with biosolid applications. Currently, there are still limited studies on microplastic accumulation in agricultural soils owing to biosolid applications (Okoffo et al. 2021). Because of the growing concern related to increased microplastic contamination in soil, it is necessary to study the effects of biosolid application rates on the microplastic

Table 1 Microplastic concentrations and polymer distributions in soil samples with previous applications of biosolids from different studies

Location	Microplastic concentration in soil samples	Polymer types	References
Melipilla, Chile	0.6 – 10.4 particles per g	PMMA, PES, nylon, LDPE, and PVC	Corradini et al. (2019)
Ontario, Canada	18 ± 22.2% – 541 ± 56.4% particles per kg	PP, PE, PES, PMMA and others	Crossman et al. (2020)
Yunnan, China (Southwestern)	7100 – 42,960 particles per kg	–	Zhang and Liu (2018)
Shanghai, China (Eastern)	78.0 ± 12.9 particles per kg (0–3 cm deep); 62.5 ± 12.97 particles per kg (3–6 cm deep)	PP (51%), PE (43%), PES (6%)	Liu et al. (2018)
Wuhan, China (Central)	2.2 × 10 ⁴ to 6.9 × 10 ⁵ particle per kg	PE (36.1%), PP (11.5%), PS (8.9%), PA (17.3%), PVC (8.5%), others (17.7%)	Zhou et al. (2019)
Shaanxi, China (North-western)	1430–3410 particles per kg	PS, PE, PP, HDPE, PVC, and PET	Ding et al. (2020)

PMMA polymethyl methacrylate, PES polyester, LDPE low-density polyethylene, PVC polyvinyl chloride, PP polypropylene, PE polyethylene, PS polystyrene, PA polyamide, HDPE high-density polyethylene, PET polyethylene terephthalate

accumulation in agricultural lands and determine how to potentially reduce microplastic accumulation in soils.

The distribution of microplastic polymers in soils is directly influenced by the quantity of microplastics present in biosolids applied to the soil. However, the soil environment is more complex than biosolids because soil receives microplastics from various industrial and agricultural activities (Okoffo et al. 2020a). Thus, it is difficult to accurately determine the extent to which biosolids contribute to microplastic distributions in soils. For instance, plastic film mulching has been reported to be a major source of microplastics (Liu et al. 2018; Zhou et al. 2019). Mulches were mainly made from polyethylene (Ahokas et al. 2014), which may have contributed to the high polyethylene abundance in the soil samples (Table 1). The proximity of sampling sites to industrial regions may also directly influence the abundance of microplastics and polymer distribution in soils (Fuller and Gautam 2016; Zhou et al. 2019). Overall, many studies have shown evidence of microplastic accumulation from biosolid land applications and microplastic polymer distribution based on local industries. Therefore, more studies are required to explore how microplastic abundance in soils may be related to land elevation and population density.

Microplastic polymer distributions in wastewater influent and effluent

The major sources of wastewater entering wastewater treatment plants include storm sewers, household sewage, and landfill leachate, which can directly affect the microplastic abundance and polymer distribution in wastewater influent and effluent (Mahon et al. 2017). Landfill leachate and stormwater runoff can be diverted to wastewater treatment plants for treatment, whereas in some countries, such as the USA, Mexico, Germany, Sweden, and Denmark, landfill leachate and stormwater runoff are discharged directly into receiving environments (Lassen et al. 2015; Smyth et al. 2021).

Figure 3A and B shows the average microplastic polymer distributions in wastewater influent and effluent samples from a study in China. The distribution patterns were similar in both samples and exhibited a high abundance of polyethylene and polypropylene. The sampling sites are located in Eastern China with a high population density, indicating high consumption of personal care products and commodities, which contributes to polyethylene and polypropylene abundance in the influent (Maddah 2016; Li et al. 2018; Sun et al. 2020). Figure 3C shows the average microplastic polymer distributions in wastewater effluent samples in Germany,

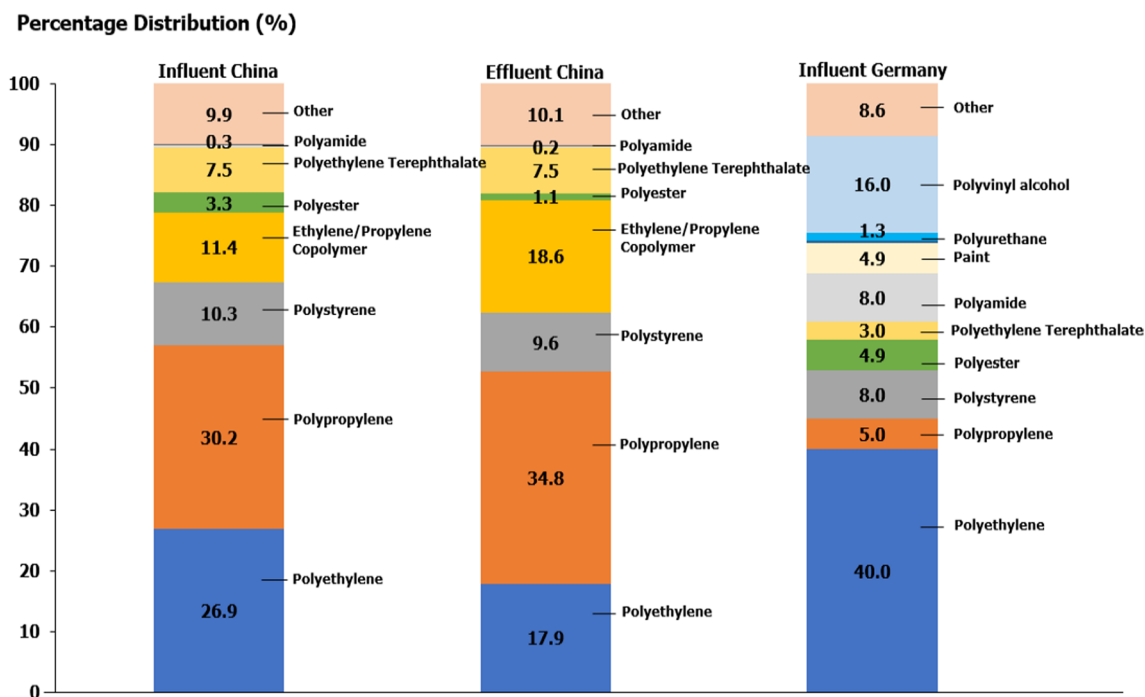


Fig. 3 The average microplastic polymer distributions in influent and effluent samples around a coastal city in China (Long et al. 2019) and effluent samples in North-western Germany (Minténig et al. 2017). It is seen that polyethylene and polypropylene are more abundant in

China, while polyethylene and polyvinyl alcohol are more abundant in Germany. The extensive usage of personal care and cosmetic products is responsible for the abundance of polyethylene polymer

which showed a significantly lower abundance of polypropylene and a higher abundance of polyamide and polyvinyl alcohol compared with the polymer distribution in China.

However, no direct evidence has been found to explain how population density or gross domestic product may affect microplastic polymer types in wastewater in different countries. The microplastic abundance in influent wastewater can also be influenced by local industries. Long et al. reported that industrial wastewater contains approximately 1.8 times more microplastics than domestic wastewater (Long et al. 2019). Overall, studies have shown that the surrounding environment and local industries close to wastewater treatment plants directly affect the microplastic compositions in wastewater influent and effluent samples.

Retention and migration of microplastics in soils

As microplastics from biosolids are introduced to agricultural soils, they may enter deeper soil layers over time and potentially contaminate groundwater (O'Connor et al. 2019). Therefore, it is essential to analyse the fate of microplastics in soil environments and understand their distribution and migration rates in different soil layers. However, owing to the complexity of soil environments, very few studies have focused on the mechanisms and rates of microplastic migration in soils (Liu et al. 2018; Zhou et al. 2020; Yang et al. 2021). Furthermore, different soil properties or external factors or both may influence microplastic migration and retention in soils (Long et al. 2019; Yang et al. 2021).

Various studies have agreed on the vertical migration of microplastics in soils (O'Connor et al. 2019; Guo et al. 2020; Wu et al. 2020; Zhou et al. 2020; Yang et al. 2021). Additionally, a few studies provided microplastic distribution profiles for different layers in soil samples, which confirmed the vertical migration of microplastics in soils (Liu et al. 2018; Crossman et al. 2020; Okoffo et al. 2021; Zhou et al. 2021). In contrast, some studies have indicated higher microplastic concentrations in the subsoils than in the topsoil, while other studies have reported opposite results (Liu et al. 2018; Crossman et al. 2020; Okoffo et al. 2021; Zhou et al. 2021). This could be explained by the different soil environments of the collected samples, which influence microplastic migration rates and result in different microplastic distributions in soils (Zhou et al. 2020). Recent studies have shown that soil characteristics, such as pH, ionic strength, and wet density, directly influence microplastic retention in soils (Hou et al. 2020; Luo et al. 2020; Wu et al. 2020). However, current studies focus on the interactions between microplastics or nanoplastic migration and different soil samples, mostly using quartz sand media as a soil profile (Hou et al. 2020; Wu et al. 2020). However, the natural soil environment is

more complicated, with varying factors and uncertainties affecting microplastic migration. Thus, further studies are needed to understand the effects of different natural soil profiles on microplastic migration behaviour and mechanisms.

Wet density

Precipitation in a region has a direct impact on soil wet density, which has been shown to be closely related to microplastic retention and migration processes (Crossman et al. 2020; Zhou et al. 2021). Crossman et al. found that greater changes in microplastic distribution were detected with higher soil wet density, indicating that microplastic retention in soils is negatively correlated with soil wet density (Crossman et al. 2020). However, the results were not obtained from controlled experiments. Therefore, other soil properties might also have affected microplastic migration. Similarly, the average microplastic abundance in the topsoil and subsoils increased by approximately 66 and 29%, respectively, because the sum of monthly precipitation increased from 450 to 1000 mm, indicating that microplastic abundance in soils is positively correlated with precipitation (Zhou et al. 2021).

The sampling sites were near a river. Therefore, the different results may be explained by flooding transferring microplastics from river beds to soils (Hurley et al. 2018a; Zhou et al. 2021). Another study showed that microplastics undergo accelerated migration in soils with a wet-dry cycle that simulates natural weather (O'Connor et al. 2019). However, the two studies may not represent the actual relationship between soil wet density and the microplastic migration process since they did not include any change in microplastic distribution to illustrate the migration of microplastic particles.

Clay and soil organic carbon concentration

Only one recent study illustrated how soil organic carbon and clay content in natural soil samples affect the microplastic migration process (Luo et al. 2020). Figure 4 shows the changes in the distribution coefficient of the solid/liquid ratio, k_d , based on different soil organic carbon and clay concentrations. According to the sorption models by Luo et al. (2020), k_d is positively correlated with the sorption capacity of soil samples, implying that soils with higher organic carbon and lower clay concentrations would have better microplastic retention. Another study demonstrated that the adsorption of carbamazepine, a hydrophobic organic pollutant, increased with increasing concentrations of dissolved organic matter (Liang et al. 2021). This result may be applied to microplastics to some extent because of the similar hydrophobic properties of microplastics and organic pollutants (Zhou et al. 2019). However, organic matter includes

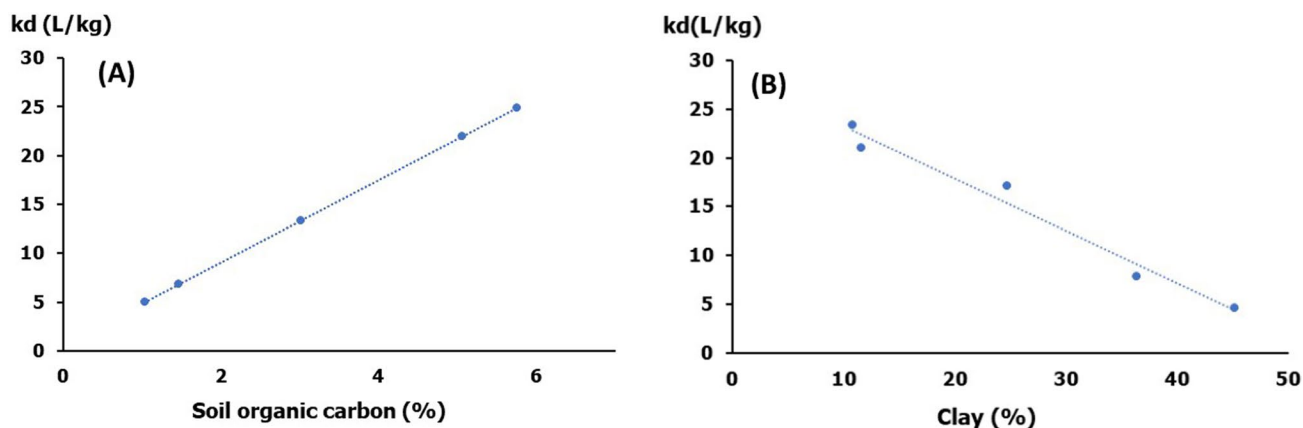


Fig. 4 Distribution coefficient values of solid/liquid ratio, k_d , with different **A** soil organic carbon concentrations and **B** clay concentrations in soil samples. k_d values are positively related to the adsorption capacity of the soil ($q = k_d * C$, where q is the soil adsorption capacity and C is the concentration) (Data source: (Luo et al. 2020)). The solid/liquid ratio, k_d , positively correlates with the adsorption capacity of soil samples, implying that soils with higher organic carbon

and lower clay concentration would have better microplastic retention. The linear regression correlation between k_d and soil organic carbon concentrations can be expressed as: $y = 4.2054x + 0.6438$ (correlation coefficient, $R^2 = 0.99$), and the linear regression correlation between k_d and clay concentrations can be expressed as: $y = -0.5375x + 28.655$ (correlation coefficient, $R^2 = 0.97$)

Table 2 Summary of pH and Fe and Al oxide concentrations of red sand, black sand, and desert sand samples (Wu et al. 2020)

Samples	pH	Fe ₂ O ₃ %	Al ₂ O ₃ %	Total oxides %
Red sand	4.97	6.57	26.9	33.47
Black sand	6.57	4.04	15.9	19.94
Desert sand	9.75	2.69	12.6	15.29

elements other than carbon. Therefore, further studies are required to investigate the effects of varying soil organic carbon concentrations under controlled experimental conditions on microplastic migration and validate the positive correlation between soil organic carbon and microplastic retention.

pH, Al and Fe oxides concentration and ionic strength

Wu et al. investigated the retention of polystyrene nanoplastics in desert sands, black sands, and red sands with different pH, soil minerals, and ionic strengths (Table 2), which may also apply to microplastics (Wu et al. 2020). The main results revealed that desert sands had the highest nanoplastic recovery at the bottom outlet of the experimental column, followed by black sands and then red sands (Wu et al. 2020), indicating that red sands with the lowest pH (4.97) have the best nanoplastic retention. In contrast, nanoplastics had the highest migration rate in desert sands, with the highest pH of 9.75. At high pH, soils become more negatively charged due to higher concentrations of OH⁻, which increases electrostatic repulsion between soil particles and nanoplastics, leading to an increased nanoplastic migration rate (Wu et al.

2020). A simulation with quartz sand confirmed the negative correlation between pH and changes in nanoplastics and soils (Wu et al. 2020). Luo et al. also studied the effect of pH on the sorption ability of five soil samples and agreed that the microplastic sorption ability of soil improves at lower pH values (Figure S2, Supplementary Material) (Luo et al. 2020).

The iron oxides in the desert sand, black sand, and red sand samples were found to be goethite (FeO₂H) and haematite (Fe₂O₃) (Wu et al. 2020). Table 2 presents the Fe and Al oxide concentrations of the three samples. The results reveal that, with significantly high Fe₂O₃ and Al₂O₃ concentrations, red sand has the best nanoplastic retention, which indicates a positive correlation between Fe and Al oxide concentrations and nanoplastic retention in soils (Wu et al. 2020). Luo et al. also found that microplastic sorption ability increased with higher concentrations of goethite and magnetite (Fe₃O₄) in soils (Luo et al. 2020). The increased sorption of microplastics occurs because Fe and Al oxides are positively charged in soils (Luo et al. 2020; Wu et al. 2020), which can attract negatively charged microplastics and nanoplastics via electrostatic interactions, lowering microplastic migration rates.

A few studies have investigated the effects of soil ionic strength on the migration of microplastics (Dong et al. 2019; Hou et al. 2020). Wu et al. reported that the ionic strength of soils is controlled by Na⁺ and Ca²⁺ ions. The recovery percentages of nanoplastics at the bottom outlet of the experimental column, with desert sand and black sand, over a range of ionic strengths in NaCl and CaCl₂ solutions are presented in Fig. 5 (Wu et al. 2020). The recovery rate of nanoplastics decreased at high ionic strengths, and black sand was found to be more sensitive

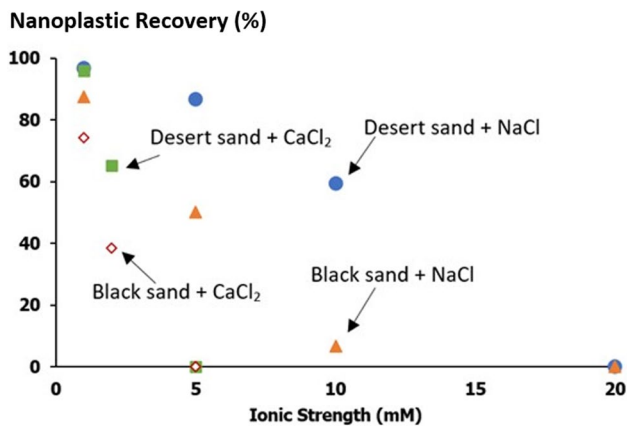


Fig. 5 Recovery percentage of nanoplastics at the bottom outlet of the experimental column over a range of soil ionic strength in NaCl and CaCl₂ solutions (Data source: (Wu et al. 2020)). Note the recovery rate of nanoplastics decreases at high ionic strength. The reduction in the recovery rate could be due to ionic strength changes in the electrostatic repulsion between nanoplastics and soils

to ionic strength changes (Wu et al. 2020). The results also indicated that nanoplastic retention in soils is positively correlated with soil ionic strength and that the nanoplastic migration rate in soils decreases as soil ionic strength decreases (Hou et al. 2020; Wu et al. 2020). This phenomenon could be explained by the reduction in electrostatic repulsion between microplastics or nanoplastics and soil particles, in which soil particles become less negatively charged as the soil ionic strength decreases.

Pore size

Several studies have found that the soil pore size has a direct impact on the microplastic migration process (Hou et al. 2020; Luo et al. 2020; Wu et al. 2020; Zhou et al. 2020). However, few studies have provided concrete results to illustrate the relationship between pore size and microplastic migration rates. Wu et al. found that nanoplastic retention is positively correlated with pore size, as soil samples with higher sand concentrations and larger pores have a higher microplastic sorption capacity (Wu et al. 2020). This result is supported by another study in which a soil sample with a looser structure and larger pores, among five samples, had the strongest sorption ability of microplastics (Luo et al. 2020). Conversely, Hou et al. found a negative correlation between pore size and microplastic retention (Hou et al. 2020). The soil structure is complicated since it includes layers and folds other than pore sizes (Luo et al. 2020). Therefore, more studies under controlled experimental conditions are needed to explore how soil texture affects microplastic migration.

Biological degradation and uptake of microplastics in soils

Microplastics can also be biologically degraded in complex soil environments, as various microorganisms utilise the carbon in plastic polymer chains for their growth (Crawford and Quinn 2017; Mohanan et al. 2020). It is necessary to investigate microplastic uptake processes in soils and the migration of microplastics in soils, to understand the fate of microplastics after entering agricultural lands via biosolids.

Microplastic uptake by microorganisms in soils

Petroleum-derived plastics, such as polypropylene, polyethylene, polystyrene, polyvinyl chloride, and polyethylene terephthalate, are the primary contributors to the formation of microplastics (Mohajerani and Karabatak 2020). However, plastic polymers undergo extremely slow biodegradation processes in soils because of their hydrophobic properties, high molecular weight, and strong chemical bonds (Hou et al. 2021; Dhaka et al. 2022). Several studies investigated the uptake of various plastic polymers by individual microorganisms isolated from soil samples, which may be used as a reference for future research (Agamuthu and Faizura 2005; Napper and Thompson 2019; Habib et al. 2020; Ru et al. 2020). A recent study estimated the toxicity of polystyrene microplastics on earthworms (*Eisenia fetida*) and found that the 50% lethal concentration (LC₅₀) of polystyrene was ~26 g/kg, which increased to ~96 g/kg after 28 days of ageing in soil (Liu et al. 2022). This indicates that the toxicity of polystyrene was reduced after ageing, possibly due to polymer degradation. However, very few studies have focused on the biological degradation process of microplastics and nanoplastics in natural soil samples because of the complexity of the soil environment.

Effects of biodegradation process on microplastic characteristics

Table 3 presents the results from various studies, showing the different biodegradation efficiencies of various microorganisms on different plastic samples based on polymer weight loss percentages or reduction in polymer molecular weight, along with other observations. Current studies show that plastics experience changes in their chemical composition, physical properties, and mechanical properties during biodegradation due to chain scissions in synthetic polymers (Napper and Thompson 2019; Ru et al. 2020; Zhang et al. 2021). Microplastic degradation efficiency is commonly determined by polymer weight losses, changes in the chemical and mechanical properties of polymers, and the

Table 3 Studies on plastic biodegradation by microorganisms from the soil environment

Plastics	Microorganism	Results	Reference
High impact PS films	<i>Pseudomonas sp.</i>	greater than 10% weight loss in 30 days	Mohan et al. (2016)
High impact PS films	<i>Bacillus sp.</i>	23% weight loss in 30 days	Mohan et al. (2016)
PS flakes	<i>Rhodococcus ruber</i>	0.5% weight loss after 4 weeks, 0.8% weight loss after 8 weeks	Mor and Sivan (2008)
Commercial PVC powder	<i>Trichocladium sp. Chaetomium sp.</i>	1.5% weight loss after 3 months, 1.62% after 6 months, polydispersity increases (chemical degradation)	Kaczmarek and Bajer (2007)
PVC films	<i>Pseudomonas citronellolis</i>	19% weight loss, about 10% reduction in molecular weight after 30 days, new functional groups	Giacomucci et al. (2019)
PVC films	<i>Phanerochaete chrysosporium</i>	11% reduction in molecular weight after 30 days, appearance of hexagonal rings in polymers	Ali et al. (2014)
PP particles (1 mm)	<i>Pseudomonas sp. ADL15</i>	17.3% weight loss after 40 days, observed C-H alkyl stretch	Habib et al. (2020)
PP particles (1 mm)	<i>Rhodococcus sp. ADL36</i>	7.3% weight loss after 40 days, observed C-H alkyl stretch	Habib et al. (2020)
Commercial PE bags	<i>Bacillus, Staphylococcus, Micrococcus, Listeria and Vibrio</i>	5% weight loss after 8 weeks	Kumar et al. (2007)
PVC wrapping materials	–	observed increase in plastic surface roughness during the 24-month period in different soil samples	Sullivan et al. (2019)*
Conventional, compostable, and biodegradable plastic bags	–	25–69% reduction in plastic tensile stress after 27 months in soil, subtle change in chemical composition	Napper and Thompson (2019)*
Shredded linear LDPE bags	<i>Serratia marcescens subsp. marcescens</i>	36% weight loss after 70 days	Azeko et al. (2015)
HDPE films	<i>Achromobacter xylosoxidans</i>	9.38% weight loss after 150 days	Kowalczyk et al. (2016)
PE-PU foams	<i>Cladosporium tenuissimum</i>	65.3% weight loss after 14 days	Álvarez-Barragán et al. (2016)
Commercial Polyether-PU foams	<i>Pseudomonas denitrificans, Pseudomonas fluorescens, Bacillus subtilis</i> and yeast <i>Yarrowia lipolytica</i>	2.8% ± 0.5% to 10.5% ± 1.5% weight loss after 5 months	Stepien et al. (2017)

PS polystyrene, PVC polyvinyl chloride, PP polypropylene, PE polyethylene, LDPE low-density polyethylene, HDPE high-density polyethylene, PU polyurethane

*Field study

production of carbon dioxide or methane during biological uptake by microorganisms (Habib et al. 2020; Mohanan et al. 2020; Zhang et al. 2021).

Since the polymer chains break during the microplastic biodegradation process, the chemical structure of the polymers or the functional groups change, and the change in the chemical composition of plastics directly affects the molecular weights of the polymers (Zhang et al. 2021). A decrease in the average molecular weight of plastic polymers has been reported in several studies (Tian et al. 2017; Ho et al. 2018; Chamas et al. 2020; Amobonye et al. 2021). Ali et al. found a reduction in molecular weight in all polyvinyl chloride samples with different microorganisms, with the most significant reduction of approximately 11% in *Phanerochaete chrysosporium* after 30 days (Ali et al. 2014).

Similarly, Giacomucci et al. reported a 10% reduction in the molecular weight of polyvinyl chloride polymers with *Pseudomonas citronellolis* after 30 days (Giacomucci et al. 2019). The change in molecular weight was determined using gel permeation chromatography in both studies (Ali et al. 2014; Giacomucci et al. 2019). These changes in the molecular weight of the polymers confirmed the biodegradation of microplastics.

Chemical structure changes and chemical bonding information have also been detected using Fourier transform infrared spectroscopy in various studies (Agamuthu and Faizura 2005; Napper and Thompson 2019; Habib et al. 2020; Ru et al. 2020). For instance, Ali et al. detected the appearance of hexagonal rings in polyvinyl chloride polymers over the course of 10 months in a soil environment,

which indicated changes in the chemical structure of plastic during biological degradation by *P. chrysosporium* (Ali et al. 2014). The mechanical properties of plastics are closely related to their chemical properties, such as polymer structure and molecular weight, and decay in mechanical strength also occurs in plastics during biodegradation in soils (Zhang et al. 2021). Subtle changes in chemical composition were detected by Napper and Thompson. However, the study reported a 25–69% reduction in tensile stress in different plastic bags over the course of 27 months in the soil (Napper and Thompson 2019).

Mechanisms of microplastic degradation

Figure 6 shows the different microplastic degradation pathways and corresponding by-products (Mahon et al. 2017; Tian et al. 2017; Ho et al. 2018; Padervand et al. 2020; Atugoda et al. 2022). Microorganisms break down polymers into smaller intermediates and then consume them to produce carbon dioxide or methane gas (Mohan et al. 2020). Thus, the weight loss of microplastics and the gas evolution of carbon dioxide and methane may be measured during the biodegradation process in soils (Zhang et al. 2021). Carbon dioxide and water are produced during the plastic biodegradation process under aerobic conditions, whereas methane, carbon dioxide, and water are produced under anaerobic conditions, with some NH_4 and organic acids (Mukesh and Anil 2005; Ho et al. 2018; Zhang et al. 2021). The production of carbon dioxide and methane is usually measured using gas chromatography or a respirometer to determine the degradation efficiency of plastics (Ho et al. 2018; Zhang

et al. 2021; Dhaka et al. 2022). However, these studies only focused on the biodegradation process of polymers in soils. Microplastics also experience other degradation processes in natural environments, including photodegradation by exposure to solar ultraviolet radiation, thermal degradation, and other catalytic degradations.

Photodegradation can occur when microplastics are located on the soil surface or in shallow soils where they are exposed to sunlight (Napper and Thompson 2019). Thermal degradation of microplastics is usually observed at temperatures above 100 °C (Chamas et al. 2020; Amobonye et al. 2021). Studies have also shown that using techniques, such as ultraviolet irradiation and transient thermal treatment, to pre-treat microplastics can increase their overall degradation efficiency (Chamas et al. 2020; Habib et al. 2020). However, when biosolids migrate to deeper soil layers, the lack of ultraviolet radiation and low temperatures result in extremely slow photodegradation and thermal degradation of microplastics (Chamas et al. 2020). Therefore, most studies on microplastic degradation in soils only consider the biological uptake of plastics (Kumar et al. 2007; Sullivan et al. 2019). Overall, the degradation efficiency of microplastics depends on several parameters, including microorganisms in soils, microplastic polymer distribution, and other potential physical and chemical processes involved.

Perspective

This review article identifies the research gap related to microplastic accumulation in agricultural soils because of biosolid land applications and the fate of microplastics in

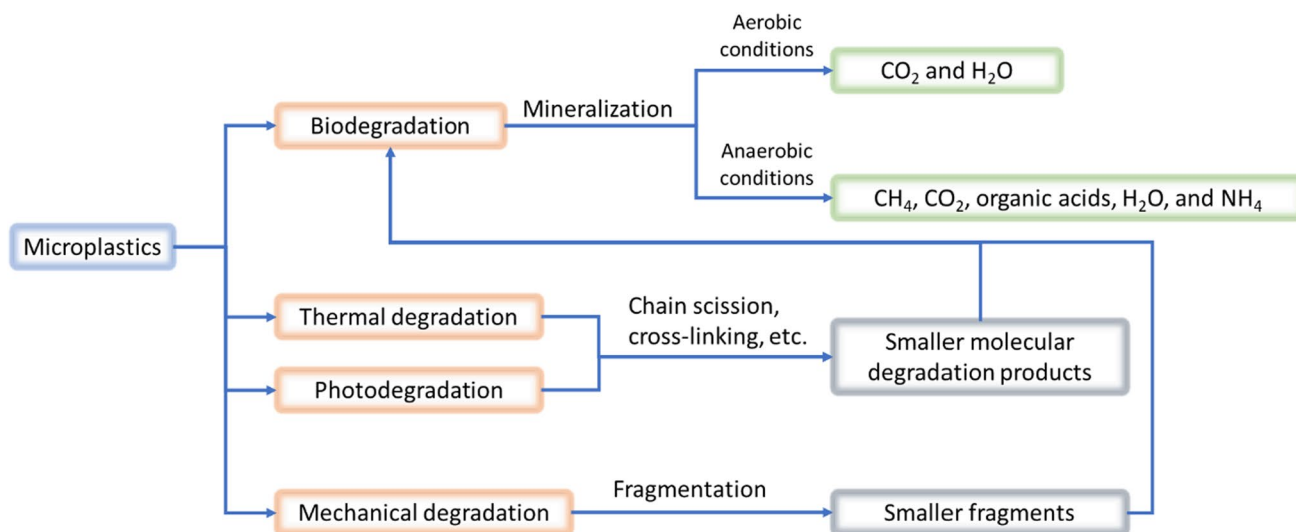


Fig. 6 Diagram of different microplastic degradation pathways and corresponding products. Microplastics undergo physical and biological changes and eventually towards gas evolution. Photodegradation

because of exposure to solar ultraviolet radiation and thermal degradation can also affect microplastics in soil

soil environments. An increasing number of studies have focused on soil environments in recent years. Their results have proven the vertical migration, biological uptake, and other interactions of microplastics with soils. However, few studies have been conducted to discuss the rate and efficiency of microplastic migration and degradation in different soils. These studies are important to understand how quickly microplastics may eventually enter the aquatic environment through soils, which is essential for the future mitigation or prevention of microplastic pollution. In addition, most current microplastic-related studies continue to focus on microplastic migration and degradation processes in aquatic environments. Therefore, more laboratory and field studies are needed to understand how soil characteristics affect the migration and retention of microplastics and nanoplastics.

Further studies on micro- and nanoplastic biodegradation, migration behaviour, and mechanisms in different natural soil profiles, which is a more complex environment with a variety of microorganisms, are recommended. Several studies have identified that biosolid applications on agricultural lands are major sources of microplastics entering the ecosystem. Thus, it is necessary to conduct more studies to understand the effects of different microplastic removal efficiencies in wastewater treatment plants, local industries, and other factors on microplastic concentrations and polymer types in the soil. Future studies focusing on understanding the mechanisms of thermal degradation, photodegradation, and horizontal migration, which are not discussed in this review, are encouraged.

Conclusion

Biosolids have been recognised as a major source of microplastics that enter and accumulate in the ecosystem through applications on agricultural lands as soil amendments or fertilisers. This review studied microplastic accumulation, migration, and degradation in soils and summarised how various factors may affect microplastic accumulation and degradation to understand the fate of microplastics in soil environments. The changes in different plastic properties during the biodegradation process by various microorganisms are also discussed, including chemical structure changes, reduction in mechanical strength and molecular weight, and carbon dioxide and methane evolution. It has been identified that microplastic content in biosolids is directly affected by local industries near sampling sites, with other factors, such as population density and elevation, which should be further investigated. Microplastics may migrate through the soil layers to enter the subsurface receptors. Based on the limited research studies, different soil characteristics may impact microplastic migration efficiency and retention in soil. Microplastic retention is positively correlated with soil

organic carbon concentration, while the concentrations of Fe and Al oxides, as well as soil ionic strength, are negatively correlated with clay concentration and soil pH. More research is needed to determine the effects of soil wet density and pore size on microplastic migration. Microplastics undergo degradation in soil environments through different pathways, including photodegradation and biodegradation. It is necessary to study how microplastic accumulation in agricultural soils can be reduced and to understand why microplastic content may differ with different removal methods used in wastewater treatment plants. These studies are essential for understanding how microplastics enter soil environments, which is crucial for implementing future policies that govern biosolid applications.

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Declarations

Conflict of interests The authors declare that they have no competing financial interests or personal relationships that could have influenced the work reported in this paper.

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