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Review

A review of the migration mechanisms of microplastics in terrestrial environments

Hui Yu[1,2], Weiqing Zhang[1,3†], Lixia Zheng[4,5], Tong Li[1], Chunxing Hai[1], Yong Wang[1], Tao Lyu[6]

[1]*College of Geographical Science, Inner Mongolia Normal University, Hohhot 010022, China*
[2]*Inner Mongolia Geology Engineering Co., Ltd. Hohhot 010010, China*
[3]*Provincial Key Laboratory of Mongolian Plateau's Climate System, Hohhot 010022, China*
[4]*Inner Mongolia Ecological Environmental Science Research Institute Co., Ltd., 010000, China*
[5]*Forensic Center for Environmental Damage of IMEPI Group, Hohhot, 010000, China*
[6]*School of Water, Energy and Environment, Cranfield University, College Road, Cranfield, Bedfordshire MK43 0AL, UK*

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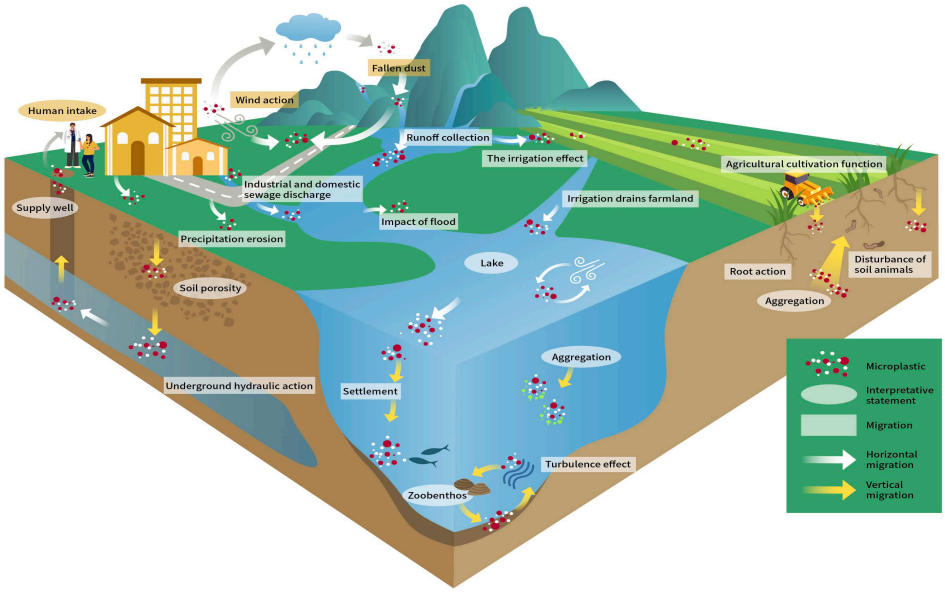
ABSTRACT

Microplastic (MP) release into the terrestrial environment has occurred since humans started manufacturing and using plastics. These tiny plastic particles can be found in various media, including the atmosphere, soil, freshwater, sediments, and organisms. MPs migrate through terrestrial environmental media due to wind, water, gravity, and biological processes. Although the variables that affect the migration process have been investigated in various settings, the mechanisms of MP migration in terrestrial environments have yet to be systematically characterized. This study classifies the migration mechanisms of MPs as physical, chemical and biological manners, and discusses the factors affecting migration mechanisms in dynamic factors, environmental factors and MP characteristics. Examining the action mechanisms of migration can establish a foundation for understanding the migration processes of MPs and provide a theoretical framework for modeling MP movement in environmental. Future research challenges include understanding the effect of MP characteristics in the migration process and simulating the migration of MPs in the environment in the long-term. Exploring the MP migration on various spatial and temporal scales, considering the life cycle of MPs is a worthy research direction.

Keywords: Mechanical migration, Microplastics, Terrestrial environments

† Corresponding author
E-mail: zwq@imnu.edu.cn
Tel: +86-1375-4096266 Fax: +86-0471-7383376
ORCID: 0000-0002-1387-364X

Graphical Abstract



1. Introduction

Microplastics (MPs) are continuing to accumulate within the natural environment as the production of plastic products increases year by year, with global plastic production increasing from approximately 1.5 million tons in 1950 to approximately 348 million tons in 2017. Of this total, 79% is discharged into the land and sea, 9% is recycled, and 12% is incinerated [1]. It has been estimated that 155-265 million tons of plastic waste will be released and discarded into the environment globally by 2060 [2]. The continuous emission and migration of MPs cause their prevalence and accumulation in various environmental media and organisms on a global scale, which causes potential ecological risks [3]. Many studies have been carried out better to understand the migration of MPs in the natural environment. MPs travel and widen the scope of pollution under the influence of wind and water. For instance, wind-borne MPs can travel far before settling on the surface through wet and dry deposition processes [4]. As a result, MPs have been found in media in unvisited areas, such as snow in the Alps [5]. Surface runoff, and rivers may be the primary route for MPs to reach the ocean [6, 7]. For instance, MPs migrate on the soil surface due to soil erosion [8], and they continue to migrate and concentrate downstream due to water action, resulting in a high abundance in estuaries [9], and enter the ocean at last. MPs in soil pores move downward due to gravity and rainwater infiltration [10]. This pathway is now a significant route for MPs to pollute groundwater [11]. Additionally, by eating, excretion, and adhesion, plankton, soil organisms, and land animals promote the mobility of MPs in the environment [12]. Research on the migration process and mechanism is the key to understanding this new pollutant, which will help us to identify pollution hotspots and develop pollution control measures.

Most early reviews evaluated the migration of MPs in individual environmental media such as seawater samples [13], soil [14], and freshwater [15], and researchers have reviewed the migration behavior of MPs in multiple environmental media such as soil–water systems [12], soil-plant systems [16], water–soil-plant systems [17], and air–soil–water systems [18, 19]. For example, Yuan et al. examined the spatial distribution of MPs in soil, water, and sediment within the middle and lower parts of the Yangtze River Basin [9]. Many studies have proposed migration models of MPs in the atmosphere, water, and soil by referring to other pollutant migration models [20, 21]. For example, Koutnik et al. examined a modeling framework for assessing the emission potential of MPs in the atmosphere, surface water, and soil [22]. These studies analyzed the migration mechanisms of MPs in individual environmental media; however, it remains unclear how MPs migrate in multiple environmental media in terrestrial environments. Analyzing and summarizing the migration behavior and dynamic internal mechanisms of MPs in multiple environments could be helpful to better understand migration trends and provide basic data to simulate the migration process at different regional scales.

Papers published between 01/01/2004 and 30/6/2023 were searched by performing a subject search (Title and Keyword) in the Web of Science database. The search specifically focused on various environmental subject areas (land/soil/terrestrial, water/freshwater/groundwater, atmosphere/wind/airborne), two plastic subject areas specific to the plastic pollutant size (nanoplastics and microplastics), and one behavior mode (transport and migration). The objectives of this review were to (1) explore the migration and internal mechanisms of MPs within terrestrial environments, and (2) propose future study directions.

2. Migration Mechanism of MPs in Terrestrial Environments

Element and compound migration in geographical environments

involves changes in spatial location, which leads to enrichment and dispersal. According to the dynamics of migration, this process can be divided into physical, chemical, and biological migration mechanical [23]. Physical mechanical refers to the migration of MPs in the environment through operational forces such as airflow and water flow. Chemical mechanical refers to the migration of MPs caused by the interaction of degradation, and adsorption in the environment. Biological mechanical refers to the migration of MPs through the processes of absorption, metabolism, growth, death, and migration of organisms; the migration and accumulation of elements in the environment through the food chain also constitute an important form of biological migration.

2.1. Physical Migration Mechanical

Physical migration mechanism refers to the horizontal and vertical migration of MPs in the atmosphere, water body, and soil environment of different scales under the action of mechanical external forces such as wind and water flow. The migration process is jointly affected by environmental factors such as meteorological conditions, hydrological characteristics, and surface characteristics, as well as MPs characteristics such as size, density, and shape.

2.1.1. Wind flow

Low-density MPs on the soil surface can be picked up by the wind and released into the atmosphere in multiple ways, including direct emission, emission from bombardment during saltation (movement of sand particles close to the surface), emission from the disintegration of large particles of aggregates as shows in Fig. 1. When MPs enter the atmosphere, wind flow cause atmospheric MPs to migrate to remote land areas and return to the surface with precipitation [24]. Atmospheric parameters (wind speed, vertical gradient, climate), surface characteristics (land use, soil texture), and MP characteristics (density, particle size, shape) affect the dispersion, migration, and sedimentation [24, 25].

Atmospheric parameters: The atmosphere parameters include wind speed, convection lift, and turbulence. Therefore, they are considered important vectors to affect microplastic transport [26]. The significant positive correlation of wind speed with MP abundance shows a role for wind speed in controlling MP transport and deposition. The minimum wind speed required for MP particles to leave the soil surface is the wind speed threshold, which is crucial to the study of wind erosion and migration. First, wind tunnel experiments showed that the wind speed threshold required for polyethylene (PE) particles and polyethylene terephthalate (PET) fibers to leave soil is 10.8 ms-1 in loam soil and 6.9 ms-1 in quartz sand [27]. As the threshold increases, the wind erosion intensity decreases nonlinearly [28].

Generally, the vertical transport of pollutants from near-ground to higher altitudes is dynamically controlled by turbulence and convection, which are highly dependent on the thermal stratification within the atmospheric boundary layer [26], as a result, MP abundance showed a decreasing trend with height, but there was a local peak in AMP abundance at 168 m. Stable atmospheric stability between 118 m and 168 m could impede the vertical dispersion of air pollutants, including MPs. On the other hand, the fine layer between 118 m and 168 m might also be associated with lower wind speeds (1 m/s at 168 m) [29].

Precipitation is an effective mechanism for removing airborne MPs, which may enter urban soils and waters [30]. Although the first rainfall exhibited the highest microplastic abundance and community diversity after long-term exposure to a dry atmospheric environment [31], there was no significant correlation between MP deposition with rainfall in this study. This may be due to rain events washing out MPs from the atmosphere before reaching the sampling sites reducing atmospheric load available for later deposition [32].

Surface characteristics: Land surface and MP characteristics affect the dispersion, migration, and sedimentation of MPs in the atmosphere [24, 25]. The key to the study of wind speed threshold is the interaction between MPs and soil particles, it is affected by soil texture and humidity. When the soil mean weight diameter

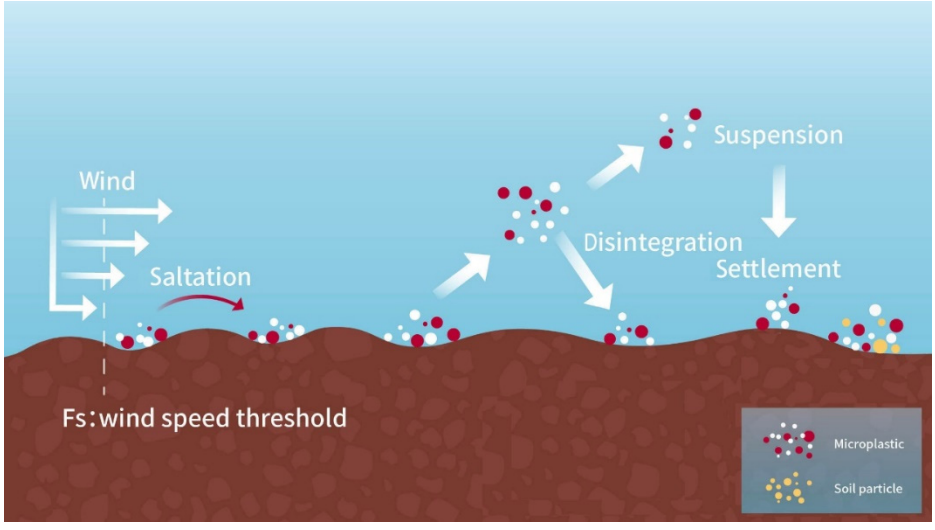


Fig. 1. Schematic diagram of the wind erosion mechanism of MPs.

(MWD) is larger than 0.3 mm, wind erosion does not significantly decrease with increasing MWD [28], indicating that large soil particles effectively prevent wind erosion of the soil surface [33], moreover, fibers are more strongly influenced than particles [34]. Wind erosion significantly increases with decreasing soil surface moisture. When the soil moisture level is less than 2%, wind erosion can be significantly inhibited [25, 28]. Although studies have demonstrated that soil moisture and soil particle size may be the main factors influencing the formation of soil–MP aggregates, the interaction mechanisms between soil particles and MPs with different particle sizes, shapes, and surface characteristics remain unclear.

Cities are often the source of MP pollution, the abundance of MPs in the fallout near the city is greater than that in the countryside, and these results support that urban areas are the main sources of MP [31]. At the same time, the existence of the underlying surface of the city will further complicate the diffusion of pollutants, and the temperature inversion layer formed over the city will hinder the diffusion of air pollutants from the ground up, resulting in the accumulation of pollutants at a certain height [35].

Wind can also cause MPs in water to escape and migrate into the atmosphere. On the ocean surface, plastic–gel aggregates formed by cation–linked bridges are ejected into the atmosphere as bubbles rise and gather on the surface [36]. Studies have found that surface circulation generated in inland surface waters due to wind can lead to the downwind migration and accumulation of MPs [37]. However, it remains unclear whether MPs escape and migrate into the atmosphere under the effect of wind in inland freshwater environments.

MP characteristics: MPs with a low density and a particle size smaller than 100 μm are most easily suspended in the atmosphere [28]. The shape also had a substantial influence on particle drift, with the fibers and films exhibiting the greatest horizontal motion [38]. It has been confirmed that the most abundant MP types in atmospheric fallout are fibers, and the length is concentrated at 200-700 μm [4], while MPs of size < 25 μm possess globally transportable features [31]. With markedly higher slopes for the spheres and cylinders as compared to the films and fibers [38], films and fibers are more susceptible to wind transport [34]. Compared with cylindrical fibers, flat fibers have a larger cross-sectional area, so the average residence time in the atmosphere is increased by more than 450%, as a result, flat fibers are much more efficient for long-distance transmission [38].

Atmospheric pollutant migration models can be used to simulate and analyze atmospheric MP emission and deposition to effectively obtain information regarding potential sources, transport trajectories, mixing, and transport distances [39]. However, MP characteristics are important factors influencing the transport of MPs in the atmosphere, and these characteristics have not been considered. Therefore, it is necessary to establish a model for atmospheric MPs to accurately determine the influences of MP size and shape. In addition, MPs in the atmosphere may comigrate with aggregated aerosol particles, which must be further investigated. The interactions between atmospheric MPs and other organic pollutants and metals in the atmosphere, their effects, and interactions on the environment, and human and ecosystem health have not been studied and need to be better understood, especially concerning nano-microplastics.

2.1.2. Water flow

MPs on the soil surface are more likely to migrate by water flow than soil mineral particles of the same size and shape. The transport processes are shown in Fig. 2. (1) In the early stages of a rainfall event, the water film that forms on smooth and gently sloping surfaces reduces the friction between MPs and soil particles due to surface runoff buoyancy, and MPs migrate with surface runoff. (2) The mesh-like flow paths formed by the scouring effect of films on rough surfaces serve as the main migration channels for MP particles. (3) When MPs enter the water, they migrate through advection, turbulence, aggregation, deposition, and re-suspension under the action of water flow. Surface characteristics (slope, roughness), Hydrological characteristics (discharge, velocity, and depth of water), and MP characteristics (density, particle size, shape) affect the dispersion, migration, and sedimentation of MPs.

Surface characteristics: The surface characteristics such as slope, and roughness determine the migration of MP particles in water. Slope and surface roughness determine the thickness and velocity of surface water flow to some extent. Increases in velocity and thickness lead to higher kinetic energy for migration [40]. For example, under 15 mm/d precipitation, the migration of PET and PE particles with sizes ranging from 0.3-1.0 mm increased with increasing slope from 5° to 25° [41]. However, when the slope was smaller, migration during rainfall weakened with increasing slope. For example, under 7.2 Lh−1 precipitation, when the slope was increased from 5° to 10°, the largest migration distance of 1215-227 μm plexiglass particles with a density of 1.18-1.19 g cm−3 decreased by 2.3 mm [40]. This may be due to the rapid decrease in film thickness due to the increase in slope under small rainfall amounts. Therefore, there may exist an MP migration threshold during slope runoff and erosion migration. This threshold is based on the thickness of the film that enables MPs to migrate under the action of buoyancy and is jointly determined by rainfall and slope. Although Nizzetto et al. established a soil MP migration model under precipitation-related erosion within a basin and determined that precipitation and the average land slope were the factors that controlled MP migration and distribution in soils [42], the model was based on theory and lacked field data support, especially regarding MP transport parameters on slopes under natural conditions. These parameters include the migration threshold, impact of land cover (e.g., vegetation), and MP sedimentation rate [43]. Currently, an effective research method to analyze the erosion and migration mechanisms of MPs is to relate these MP processes to the erosion mechanisms of soil and soil organic carbon. However, studies have demonstrated that MP migration is notable in sandy soils, which are not easily eroded, and the soil erosion amount is usually closely related to the migration of MPs with large soil particle sizes, which are easily eroded [8]. Therefore, MP and soil erosion characteristics vary, and the joint migration mechanism of MPs and soil under soil erosion conditions should be studied.

The surface mesh-like flow paths produced on rough surfaces due to precipitation are the main channels for MP migration on slopes. The surface of the natural environment is rough and uneven.

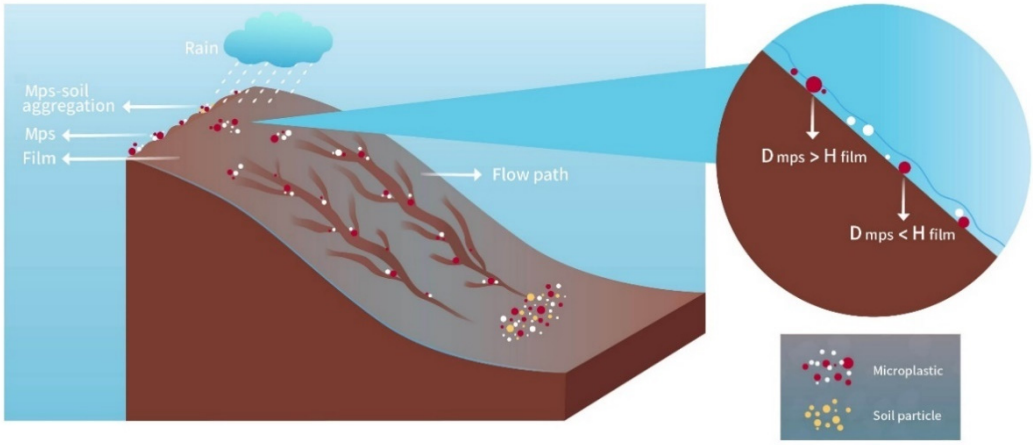


Fig. 2. Schematic diagram of the hydraulic erosion mechanism of MPs.

On the one hand, a rough surface increases friction during MP migration and inhibits migration. On the other hand, rough surfaces easily produce narrow flow paths and promote migration. Studies have shown that rough surfaces may increase MP particle friction at the early stages of precipitation [40], however, with continued rainfall, flow paths are more likely to form, which increases the thickness of the local runoff layer and promotes MP migration. For example, under the same rainfall conditions, the concentration of high-density polyethylene (HDPE)-MP particles in runoff erosion sediments in coarse loamy sand is higher than that in silty loam [8], which may occur because flow paths are more likely to form in loamy sand soils. In addition, the impact of uneven natural precipitation and raindrops could easily result in the formation of flow paths and could increase the thickness of local runoff layers to promote the migration of MPs [40]. For example, unevenly distributed natural rainfall with an intensity of 3.10 mm/d resulted in higher MP migration than uniform indoor simulated rainfall amounting to 5 mm/d [41].

Hydrological characteristics: Hydraulic transport is a process in which MPs migrate with runoff through advection, turbulence, aggregation, deposition, and resuspension, and this process is influenced by hydrological characteristics such as the discharge, velocity, and depth of water, and MP characteristics [44, 45]. High water velocity facilitated the MPs to be migrated for a longer distance, and high-water flow was conducive to transporting more MPs from source points [46]. In water bodies with low flow velocity and deep rivers or lakes, low-density MPs tend to stay at the surface, and their transport is mainly controlled by advection-driven migration and sedimentation, while high-density MPs tend to aggregate and settle into sediments [47]. In contrast, high flow velocity tends to cause turbulence in water bodies, which promotes MP migration along the vertical and horizontal directions. Compared to that at low flow velocity, the concentration of MPs in water bodies increases by 3-4 times at high flow velocity [48, 49]. 10 times as much plastic was transported in the Seine during high discharge periods as during low discharge periods [50]. Additionally, flood events can even resuspend up to 70% of MPs deposited in riverbeds and cause MP migration downstream [51]. However, the change in flow velocity is not only related to the discharge but also affected by the topographic height difference of the river bed. For instance, the stream-wise velocity was nearly zero at the riverbed due to bottom friction, and it increased with elevation by reaching the maximum of 0.4 m/s near the river surface [52]. Therefore, the relationship between flow velocity and MP concentration was different from that between discharge and concentration [46]. Most studies give qualitative overviews on MP contamination in rivers and do not go further towards quantification [52], variability within study sites shows a large heterogeneity indicating that generalizations are difficult, as a result, insufficient knowledge of processes of MP transport limits the definition of reliable standards [53].

Numerical modeling has been recognized as an important tool in predicting the fate and transport of MPs in aquatic systems [54], such as Lagrangian particle tracking algorithms and Eulerian models [55, 56]. Geng et al. coupled river hydrodynamics and particle aggregation-breakage kinetics to multiple interactive transport equations [52]. Impacts of the aggregation-breakage process as well as spatial variation of diffusivity on the fate and transport of MPs in the river system were quantified. However, degradation and fragmentation are not considered, which are influenced by other various factors, such as biofilm, heavy clay, zeta potentials, dissolved organic matters, and ionic strength. In this simulation, the influence of flood events, and riverbed and riparian vegetation on MP transport were not considered. In the future, the permanent monitoring of distinct river sections and the characterization of MP budgets ought to be considered.

MP characteristics: The characteristics of MPs, such as their size, shape, and density, affect the migration process during migration. The interactions between MPs and soil particles, which are related to MP particle size and soil moisture, inhibit erosion and migration on the soil surface. Zhang et al. found that the migration of medium-sized MPs (0.3-1 mm) was higher than that of small MPs (< 0.3 mm) and large MPs (> 1 mm) under precipitation-driven erosion conditions [41]. Rehm et al. reported that 250-300 μm MPs more readily migrated than 53-100 μm MPs [8]. Nizzetto et al. determined that when the MP particle size is smaller than 100 μm, MP particles are more likely to form stable aggregates with other particles and do not easily migrate [42]. In conclusion,

MPs with a particle size smaller than 0.3 mm are more likely to aggregate with soil particles to form stable aggregates, while those with a larger particle size are less stable. In addition, over time, the proportion of aggregates formed by MPs and soil increases [8, 57]. Moreover, the erosion rate of MPs in dry soil particles is higher than that in wet soil, irrespective of their particle size [8]. This indicates that dry and loose soil surfaces can be easily eroded, while additional binding forces might exist between MPs and mineral particles in moist soils, leading to a decrease in the MP concentration in eroded soil. Although previous studies have focused on the impact of the interaction between MPs and soil particles on erosion and migration processes, the mechanisms involved in these interactions remain unclear. The diameter of large MP particles is positively correlated with their settling and rising velocity [53], while the sedimentation of smaller plastic particles depends on their aggregation with suspended particles. Regarding MP particles between 2 μm and 10 mm in diameter, the sedimentation rate increases with increasing diameter, while regarding particles between 0.1 μm and 2 μm in diameter, the sedimentation rate decreases with increasing diameter [44, 58], however, it was shown that the interaction between particles is limited to small MPs whereas particle interactions have no significant effect on larger (> 0.2 μm) MPs and particulate matter [59, 60]. Therefore, the effect of particle size on MP aggregation should be studied further.

The shape of MPs has a significant impact on their migration. MPs with an irregular shape have greater buoyancy than those with a spherical shape. Meanwhile, although most of the fibers have been removed by the primary and secondary treatments in the sewage treatment process, due to the property that fibers could escape from filters or membranes more easily, their relative abundance would increase in the final effluent [61]. As a result, films and fibers are less likely to settle, leading to a wider distribution and higher abundance of fibers in both air and water [24, 62]. In river sediments, the main MPs larger than 300 μm are fibers, while MPs smaller than 300 μm are mainly fragments [63]. However, the impact of shape on the settling of MPs may be dependent on their size, and this correlation may only become apparent once the particles reach a certain size [64]. The sedimentation rate of debris-, fiber- and particle-shaped MPs with particle sizes between 1 and 3 mm shows a decreasing trend [58]. Irregular particles sink and rise more slowly than spheres of the same size, as secondary movements reduce their velocity [62]. Moreover, the shape effect becomes apparent only when MP particles reach a certain size [65]. Large and irregularly shaped MP particles are more susceptible to turbulence and flow migration and can be carried by the flow, while smaller, smoother, and rounder particles more often appear at the surface and in the stratosphere [48]. Therefore, in stream sediments, MPs larger than 300 μm mostly comprise fibers, while those smaller than 300 μm mainly comprise debris [63]. MPs that are denser than water tend to settle [66], while low-density MPs are abundant in surface waters, and sediment mostly comprises dense MPs [67]. According to one study, denser particles settle much more easily than water, leading to increased concentrations of PE and PP particles and a decreased concentration of PET particles in most bodies of water. Specifically, water bodies exhibit elevated levels of PE and PP particles due to their density, whereas the PET content is usually low [66, 68]. An MP concentration gradient with depth was found in rivers with a low flow rate and great depth: the concentration of low-density MPs decreased from the surface to the sediments, while high-density MPs showed the opposite gradient [69]. However, due to aggregation between MPs and other particles, such as microorganisms, metal ions, and organic matter, when MPs enter inland waters [44, 63], the MP density and size increase thus promoting settling [70]. For example, the adhesion of microorganisms (such as algae) and the adsorption of solid particles resulted in an increase in the density of polypropylene (PP) aggregates from 0.9 to 1.19 g/cm^3, which was much higher than the initial density. The formation of biofilms increases the sedimentation rate of MPs [58, 71]. It was experimentally found that the maximum migration depth of low-density PE particles is greater than that of PET particles in soil [72]. Furthermore, the adsorption of other components onto MPs alters their surface characteristics, thus affecting their migration behavior [67, 71]. For example, PP particles contain substantial amounts of adsorbed surfactant molecules due to their large surface area. Therefore, PP exhibits a higher migration capacity than PE [73].

Because MPs of different shapes, densities, and sizes exhibit distinct migration behaviors in water environments, the universality of most MP migration models is low. Therefore, the influence of MP characteristics on migration mechanisms must be studied further. The influence of river morphology, tidal flow, dams, and vegetation on MP migration in rivers should be more closely examined. In addition, the characteristics of MP particles can change, with residence time in the environment due to aggregation, biofilm development, degradation, and flocculation mechanisms [46], however, the changes in the migration process caused by these processes need to be further studied.

2.2. Chemical Migration Mechanisms

2.2.1. Aggregation and precipitation

Microorganisms, metal oxides, dissolved organic matter, and clay minerals can be found in natural water and soil. These elements can combine with MPs through various interactions such as van der Waals forces, electrostatic repulsion forces, hydrogen bonding, and π-π interactions as shown in Table 1 [70, 74]. The resulting aggregates settle into sediments, but they can be dispersed in the water body by turbulent flow and benthic organisms. This process is one of the main mechanisms of MP migration, especially in oceans and other large water bodies [70]. MPs undergo significant aggregation and breakage as they are transported downstream by river flows. Aggregation occurs near the surface at the early stage of the release of MPs in the river, while the process becomes limited as the microplastic plume is gradually dispersed and diluted downstream [52]. In soil, aggregate formation also occurs between MPs and soil particles, especially for MPs with a smaller particle size, which are more likely to form stable soil–MP aggregates [8, 57]. Although MP migration, suspension, and sedimentation in the atmosphere can be affected by the aggregate effects of MPs, this effect has not been generally documented [20].

At present, most studies focus on the aggregation and synergistic migration of MPs with heavy metal pollutants, organic pollutants,

Table 1. MPs transport mechanisms in sand and soil

Porous medium (mm)	pH	Organic substance	Mineral	Electrolyte (mM)	MP size (μm)	MP polymers	MP shape	Main conclusion	Sources
Quartz sand average diameter:0.60 mm	N/A	Naphthalene	N/A	NaCl: 0.5, 5, 50	0.12	PS	Spheres	Naphthalene reduced PSNP migration in sand columns, and PSNP decreased with increasing of IS.	[83]
Glass beads diameter: 0.707–0.841 mm.	6.5-7.5	TC: 0, 1, 5, 10, 120 mg/L	N/A	KCl, $CaCl_2$: 1, 10, 50, and 100	1.0	PS	Spheres	The presence of TC slightly inhibited PSMP mobility in K^+ solutions but facilitated it in Ca^{2+} solutions.	[76]
Quartz sand: 0.3~0.425 mm	6	Colloidal biochar	N/A	NaCl: 5, 25	0.02, 0.2, 2	PS	Spheres	Biochar decreased MP transport at both low (5 mM) and high (25 mM) IS levels; heteroaggregates drove the decreased transportation.	[87]
1:1 mixture of soil and coconut fiber: 0.85 mm	N/A	200 mg NH_4NO_3 (nutrient)	40 mg $CdCl_2$	water	100–200 and <100	PP	N/A	Springtails exhibited a notable ability to transfer MPs into the soil. Adsorbed nutrients, the target pollutant (cadmium; Cd), and green fluorescent Escherichia coli were also transferred with the MPs.	[88]
Quartz sand: average size 0.337 mm	7.5	DEHP	GT:304.2 ± 5.8 nm	N/A	0.05	PS	Spheres	GT decreased the transport of NPs, and the presence of DEHP decreased it further. The deposition of NPs due to chemical heterogeneity was more significant than that due to the formation of chemical bonds and van der Waals, electrostatic, and hydrogen interactions.	[89]
Quartz sand: 0.3-0.425 mm	6	N/A	GT, HT	NaCl: 5, 25	0.02, 0.2, 2	PS	Spheres	GT or HT did not change the transport and deposition of the smallest plastic particles (0.02 μm)	[90]
Quartz sand: 0.3~0.425 mm	6±0.1	N/A	Kaolinite	NaCl: 5, 25	<1.0	PS	Spheres	The presence of kaolinite increased the transport of CMPs but did not affect the transport behavior of AMPs.	[91]
Quartz sand: 0.3~0.425 mm	6	N/A	N/A	NaCl: 5, 25	1.0	PS	Spheres	The presence of bacteria decreased CMP transport in sand columns. In contrast, the presence of bacteria increased AMP transport but decreased their deposition in sand columns under both IS conditions.	[85]
Quartz sand average diameter: 0.416 to 0.6 mm	6	BSA, trypsin	N/A	NaCl: 5, 25	0.2	PS	Spheres	The presence of BSA increased the transport of both CMPs and AMPs, while the presence of trypsin decreased the transport of CMPs yet increased the transport of AMPs in porous media.	[92]
Quartz sand average diameter: 0.85-2.0 mm	4.0, 7.0, 9.0	N/A	N/A	KCl: 0.1, 0.01, 0.001	N/A	PE, PP	Spheres	The mobility of PE and PP under CTAB conditions was much higher than that under SDBS conditions, and the mobility of PP was greater than that of PE under all conditions considered.	[73]
Natural soil: sandy soil and clay loam soil: 0.6-0.7 mm	6.7:4.76	N/A	N/A	NaCl: 5 mg L^{-1}	1.0	PS	Spheres	Surface functional groups increased electrostatic repulsion between aged MPs and soil particles. Higher transport of MPs and aged MPs occurred in sandy soil than in clay loam soil.	[10]
Sand:0.36 mm, gravel: 2-8 mm	N/A	Wastewater: organic matter	N/A	N/A	45-53	PS, HDPE, PP, PE	Beads	There was no significant difference in the distribution of PP, HDPE, and PS particles with or without biofilms but the infiltration concentration of microbeads with smaller particle size was higher.	[93]

Porous medium (mm)	pH	Organic substance	Mineral	Electrolyte (mM)	MP size (μm)	MP polymers	MP shape	Main conclusion	Sources
Natural sea sand: 0.45±0.03 mm	N/A	Suwannee River humic acid	N/A	Artificial seawater with a salinity of 35 PSU, 3.5 PSU	0.2	PS	Beads	The transport rate of NPA^- decreased over time due to the high attraction between NPs and deposited NPs; NPA^+ deposited due to the electrostatic attraction between NPA+ and negatively charged sand.	[94]
Quartz sand average diameter: 0.35-0.45 mm	4.0,6.0, 8.5	HA	N/A	NaCl, KCl: 1, 10, 50 $MgCl_2$, $CaCl_2$: 0.1, 1, 10	0.25-5	PET	Fragment	PET MP transport increased with decreasing electrolyte concentration and valence and increasing pH and HA concentration.	[95]
Spherical glass beads: 0.707-0.841 mm	N/A	HA: 0, 5, 10, 20 $mg \cdot L^{-1}$	N/A	KCl: 1, 100	1.0, 0.2	PS	Spheres	The transport behavior of smaller CMPs (0.2 μm) was more sensitive to changes in IS and the presence of HA than that of larger CMPs (1.0 μm).	[82]
Fine-grained seafloor sand: d50=0.45±0.03 mm	N/A	N/A	N/A	N/A	2.0, 1.5, 0.8, 0.6, 0.4, 0.1	PS	Spheres	The transport of MPs increased with decreasing MP size (2.0-0.8 μm), but the smaller MPs (0.6-0.1μm) showed stronger inhibition of mobility.	[96]
Quartz sand: 0.425-0.6 mm, loamy sand soil: 0.25 mm	7.5±0.5	HA:1, 5, 25 μg/ml	N/A	$Cacl_2$: 2.5, 0.45, 0.9	NPs: 0.19, 0.11, 0.05 NPC, NPA: 0.048, 0.051	PS	N/A	The mobility depended on NP properties such as the size, surface electrical charge, and surface reactivity, as well as on matrix properties such as the type of medium and its composition. Exposure to the environment may change NP surface properties, which may in turn affect their mobility.	[81]
Quartz sand average diameter: 0.26 mm	N/A	N/A	(R) GO	NaCl: 10, 30, $CaCl_2$: 0.1, 0.5	0.1	PS	Spheres	In Na^+saturated porous media, PSNPs preferred to interact with (R) GO breaking through the sand column. In Ca^{2+}saturated porous media, the transport of both (R) GO and PSNPs was inhibited.	[97]
Sand: 0.6–2 mm, 0.21–0.60 mm, and <0.21 mm	N/A	HA	N/A	N/A	10-450	PP, PA, PE, PET	N/A	Surface hydrophobicity showed a strong positive correlation with MP mobility and the penetration depth of MPs increased with decreasing MP particle size and increasing sand diameter.	[72]
Quartz sand: d50=383 mm	N/A	N/A	N/A	N/A	21-535	PE, PP	Fine, coarse	When MPs were subjected to larger numbers of wet-dry cycles, the penetration depth significantly increased.	[98]
Quartz sand: d50=0.265 mm	6.0±0.2	Suwannee River NOM	N/A	NaCl: 100	0.028	PS	Beads	Exposure to 10 FT cycles led to aggregation and reduced mobility, especially at low IS in the absence of NOM.	[99]

polystyrene nanoplastics (PSNPs); humic acid (HA); ionic strength (IS); graphene oxide (GO); reduced graphene oxide (RGO); cetyltrimethylammonium bromide cationic surfactant (CTAB); sodium dodecylbenzenesulfonate anionic surfactant (SDBS); bovine serum albumin (BSA); amine modified polystyrene (AMP); carboxylate modified polystyrene (CMP); tetracycline (TC); extracellular polymeric substances (EPS); natural organic matter (NOM); freeze thaw (FT); practical salinity units (PSU); nanopolystyrene (NP); nanopolystyrene functionalized with surface carboxyl groups (NPC); nanopolystyrene functionalized with surface sulfonic groups (NPS); nanopolystyrene functionalized with surface low-density amino groups (NPA); negatively charged NPA (NPA-); positively charged NPA (NPA+); fulvic acid (FA); diethylhexyl phthalate (DEHP); goethite (GT); hematite (HT).

and organisms [75, 76]. It is generally accepted that heavy metal ions can be adsorbed at equipotential points on the surface of MPs via electrostatic interactions and complexation [77]. Various forms of arsenic can be adsorbed onto PE MPs, and the adsorption behavior depends on the PE structural properties, arsenic morphology, and water chemistry [78]. Various organic pollutants, such as polycyclic aromatic hydrocarbons (PAHs), polychlorinated biphenyls (PCBs), and dichlorodiphenyl trichloroethane (DDT), can be adsorbed onto MPs [79]. Different microbial communities can be adsorbed onto the surface of marine plastic debris (PP and PE) [80], which is an important factor leading to MP aggregation. The heterogeneous aggregation of PP MPs was estimated to affect approximately 50% of MPs, and microalgae accounted for 50% of the aggregates [71, 76]. Once these toxins are leached/desorbed from MPs, they may pose severe risks to microorganisms in biofilms [61]. Therefore, future research should attach much importance to the effects of synergistic between pollutants and MPs on migration processes.

The aggregation–precipitation process is also affected by both the physical and chemical properties of the surrounding environment, as well as the characteristics of the MPs themselves. Factors such as the ionic strength (IS) and valence state can increase the precipitation of MPs, while decreases in pH and HA concentration can also have an effect. However, it has been observed that the characteristics of the MPs themselves also play a role in this process. Specifically, smaller particles (0.2 μm) are more sensitive than larger particles (1.0 μm) to changes in the IS and HA of the surrounding medium [81, 82].

The overall effect of aggregation on MP migration processes is a combination of inhibitory and promoting effects [76]. On the one hand, aggregation promotes precipitation by increasing the particle density and size and changing surface characteristics [70], for example, the adsorption of the organic pollutant naphthalene on the surface of MPs increases the zeta potential and reduces the energy barrier with sand, which is conducive to the retention of MPs in sand columns [83], additionally, extracellular polymeric substances (EPS) can form hydrogen bonds with plastic particles to promote deposition [84]. On the other hand, negatively charged MP-bacterial aggregates increase the repulsive forces with the medium surface, which promotes MP migration [85]. The influence of aggregation on migration and sedimentation may be the combined result of various mechanisms, but more quantitative research is needed to understand the degree of influence of different mechanisms. In addition, with increasing residence time in soil, MPs impose greater effects on the accumulation of soil particulate matter. Under simulated rainfall conditions, the MP and soil particle accumulation rate reached 67% after 1.5 years, twice as high as that during the initial period [86]. However, due to the short study period in most recent studies, the inhibitory effect of aggregation on migration is not significant, and long-term quantitative simulation studies are needed.

2.2.2. Degradation

The breakdown of MP in the environment is a result of both abiotic and biotic processes. Abiotic degradation is the disintegration of MP through photodegradation, thermal degradation, chemical oxidation, and physical wear, while MPs are broken down by living things like bacteria, fungi, and other microorganisms during a process known as biotic degradation. The surface mechanical properties of MPs are destroyed through abiotic and biotic processes, which results in long-chain fracturing of polymers and a reduction in molecular weight, and the MPs are eventually transformed into CO2 and H2O [10, 100]. In the degradation process, cracks and voids emerge on the surface of MPs, which become rough, and the specific surface area of MPs increases, in addition, functional groups and the biofilms formed through oxidation [101, 102]. These changes affect the adsorption, aggregation, and migration behaviors of MPs by hydrophobic interactions, π-π bond interactions, electrostatic interactions, biofilms, and other factors as shown in Fig. 3 [10, 103].

In the presence of light, MPs absorb energy, transitioning into an excited state that triggers the cleavage of some chemical bonds within the plastic molecules [104], especially under UV induction, which is considered to be an important process of hydrocarbon polymer aging [105]. UV light irradiation can induce yellowing, cracking, and even pronounced fragmentation of MPs, which influence their adsorption capacity and hydrophobic bonds [104]. Due to UV aging, the adsorption capacity of Cu^{2+} on ethylene glycol phthalate increased from 51.2 μg/g to 172.8 μg/g [77]. Thermal degradation parallels photodegradation, as both processes are forms of oxidative degradation. Both mechanisms rely on the weathering by free radicals, leading to chain breakage and the generation of oxygenated intermediates. Physical degradation is induced by various elements such as wind, waves, and human activities. This process increases the number of MPs, reduces their size, and even generates nano plastics, however, Physical degradation has a limited effect on microplastic degradation, for example, the original PP generated only 10.7 ± 0.7 particles under 2 months of mechanical wear, while the UV-aged (12 months) PP yielded 6084 ± 1061 particles after 2 months of mechanical wear [106].

Plastic degradation consists of three major steps, colonization, deterioration-fragmentation, and mineralization Bacteria, fungi, and other microorganisms decrease the polymer's molecular weight, which facilitates the transit of molecules via the cell membrane and enzymes inside the cell to destroy it more easily [105]. studies have identified specific plastic-degrading microbial strains and elucidated their biodegradation pathways [104]. For instance, two bacterial strains Rhodococcus sp. strain 36 and Bacillus sp. strain 27 could degrade PP, after forty days, and the percentage of weight loss was 4.0%, and 6.4% [107].

Variable efficacy of MP degradation depends on the type of plastic (size, crystallinity, salinity) and environmental factors, such as oxygen, and temperature [108]. In general, the thin layer is more easily degraded than the bulk plastic in this form of degradation due to the usual development of cracks and fusion with other cracks [105], the smaller the MP size with greater surface area, which offers a larger reactive region for MPs. However, further studies have shown thermal stability of MPs decreases as the size is reduced to the nanoscale. As a result, MPs are not solely dependent on the specific surface area, but also on the crystallinity and chain densification that accompany size reduction size reduction to the nanoscale [104]. Higher crystallinity reduces the elasticity of MPs, making them more prone to breakage [109], while the

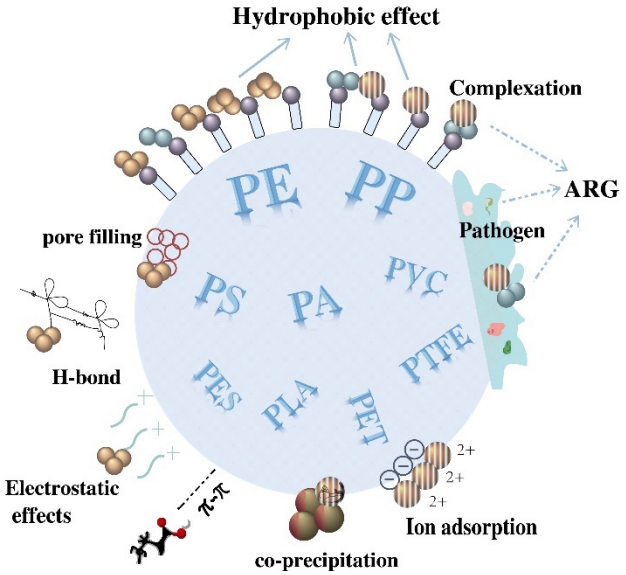


Fig. 3. Physical-chemical mechanisms on the surface of MPs [114].

amorphous regions are more accessible to microorganisms than the crystalline regions [110]. Oxygen availability and high temperatures can expedite the aging of MPs [111, 112], so water might decelerate the aging of MPs compared to exposure to air [111].

MPs may be broken down into smaller molecules by living organisms without producing any toxic byproducts, but these smaller particles may nevertheless linger in the environment and may even be consumed by creatures, having subsequent consequences on the health of those animals and the dynamics of the ecosystem [105], especially the aged MPs improved arsenic adsorption amount to the level of pure soil due to newly generated O-containing functional groups [113]. Therefore, the adsorption of pollutants by aging MPs has become the focus of future research.

In the degradation process, cracks and voids emerge on the surface of MPs, which become rough, and the specific surface area of MPs increases. The rough surface of degraded MPs can provide a large number of available adsorption sites [101]. Studies have verified that compared to of low-density polyethylene (LDPE) in the initial state, aged LDPE exhibits an increased surface area that takes longer to reach heavy metal adsorption equilibrium [115]. The functional groups formed through oxidation on the surface of MPs and the biofilms formed by biological adhesion [101, 102] affect the surface morphology, hydrophilicity, chemical composition, and structure of MPs. The adsorption amount of Cd(II) on aged PE increased the adsorption capacity by 4.7% relative to that on pristine PE, in which the oxygen-containing functional groups of APE could provide binding sites to increase the adsorption [116]. The carbon groups of MPs can combine with the oxygen atoms of water to form hydrogen bonds, which enhances their hydrophilicity. During the photodegradation of polystyrene MP particles (PS-MPs), oxidative functional groups can be formed to increase the carbon index [117, 118], promote MP migration in porous media [119, 120], enhance the adsorption capacity of hydrophilic substances, such as pesticides and antibiotics [114], and inhibit the adsorption of hydrophobic organic pollutants [121, 122]. Biofilms change the surface charge of MPs, thus affecting the electrostatic effects encountered in MP adsorption and aggregation processes. Compared to MPs in the original state, ultraviolet-irradiated MPs exhibit higher a negative charge, which increases electrostatic repulsion from soil particles and promotes MP migration in porous media [10]. Although researchers have analyzed the adsorption, aggregation, and migration behaviors of MPs under different degradation modes, aging exerts different effects on behaviors in MPs with different characteristics [123] and further investigations of the different action mechanisms and synergistic effects should be conducted. For example, further research is needed to better understand the adsorption mechanisms of microorganisms on MPs and whether these mechanisms are related to the degradation of MPs [103].

2.3. Biological Migration Mechanism

Biological migration refers to the migration of substances due to biological processes such as absorption and metabolism. Plant, animal, and human activities in the terrestrial environment influence these migration processes. On the one hand, organisms absorb MPs from surrounding media through metabolic life processes [124], and MPs migrate in environmental media through the processes of feeding and excretion. MPs can migrate through the food chain and participate in biochemical recycling through a wide range of migration mechanisms as shown in Fig. 4. On the other hand, biological factors influence the migration and deposition process of MPs by altering the conditions of the surrounding media.

2.3.1. Direct effect of biological metabolism

Plankton, fish [125], soil organisms, etc., promote the migration of MPs in soil and water environments through feeding, excretion, adhesion, and other activities [12]. For example, plankton ingested MPs and migrated from the surface to the deep water [126], the plankton filter MPs from the water column and pack them into fecal particles that sink to the bottom, which can lead to the vertical migration of MPs. The fish can increase MP concentration in high-suitability habitats 1–3 times larger than that in low-suitability habitats, which can lead to the horizontal migration of MPs [125]. Pristine MPs (250–300 μm) are broken down into smaller particles (256 nm and 1.4 μm) when ingested by earthworms [127], and some particles may remain in the digestive tract of organisms for long periods, while some particles may even enter various organs and tissues through the walls of digestive tracts, most MPs are returned to the environment through the excretion of feces [128, 129]. Mammals, fish, and invertebrates have been found immobilizing MPs on the organism surface. For example, large amounts of plastic debris had been found in sperm whales stranded [130]. The contents of MPs in the skins and gills of fish approached or even exceeded those in the guts [131]. The bioturbation of infauna can lead to the migration of MPs to the deep layer, and the resuspension of deposited MPs. A study found that 8% of MPs on the sediment surface were transported to the deep layer after three weeks of bioturbation [132]. However, the migration and cumulation of MPs in different aquatic organisms and related physiological processes is still a difficult problem, for example, the impact of bioturbation on the overall migration direction and

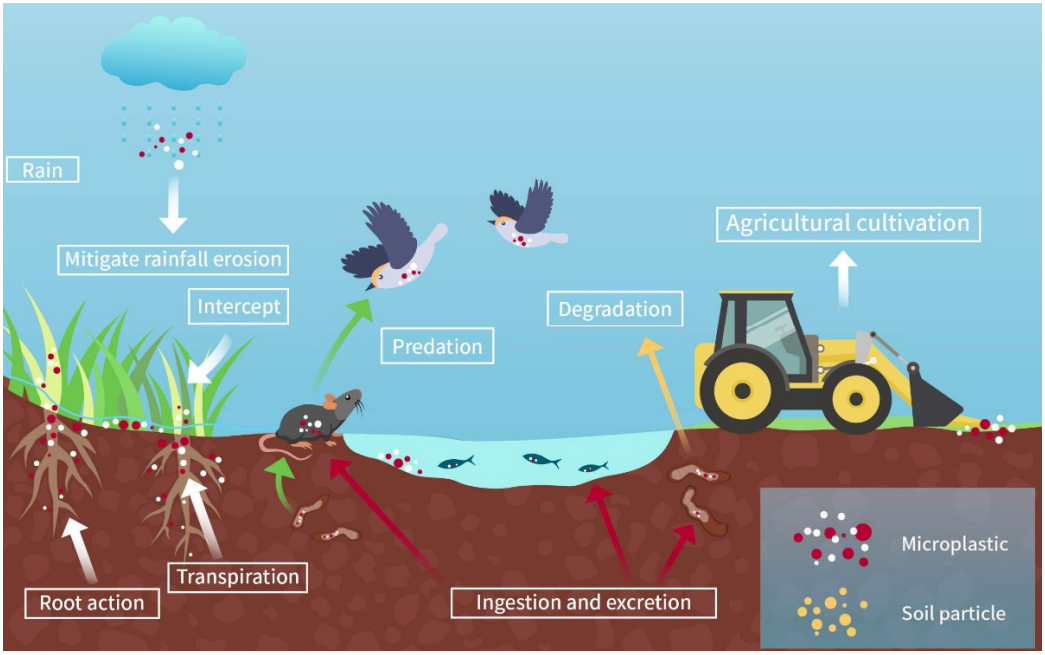


Fig. 4. Schematic diagram of the biological migration mechanism of MPs.

flux of MPs in sediments must be further investigated [133].

Laboratory studies have demonstrated that MPs transferred along trophic levels from Artemia nauplii to zebrafish (Danio rerio) [134], from mussels (Mytulis edulis) to crabs (Carcinus maenus) [135]. Despite the mounting evidence of MPs within all levels of freshwater food webs described above, it is important to note that the uncertainty of whether MPs in different tissues of fish undergo biomagnification or biodilution is controversial [125]. For example, comparing MP concentrations in whole organisms, it may be misleading to analyze only the quantity of MPs in specific tissues. The time scales that MPs are retained in tissues should be considered.

In most studies, plastics were detected in the gills and guts of fish were fibers and smaller than 5 mm [136], the selectivity of MP uptake was dependent on the particle resembled natural prey, for example, benthic sea turtles showed strong selectivity for soft, clear plastic, suggesting that sea turtles ingested plastic because it such as jellyfish [137]. Studies have shown that fish selectively ingest colored MPs, with some fish tending to ingest blue MPs [138], however, this may be because blue MPs are the most common type of MPs originating from trawls and other fishing gear [136]. Therefore, the characteristics of MPs and the habitat and feeding type of different organisms affect the transfer of MPs in the food chain, it should be considered in the future.

MPs have been detected in soil organisms, terrestrial wildlife, and humans and may be transferred and accumulated along trophic levels [139]. Nanoplastic particles can enter plant root systems through cracks in the lateral roots of plants and migrate from the roots to shoots through transpiration [124]. Nevertheless, there remains a lack of empirical studies verifying the transfer of MPs along trophic levels in organisms in terrestrial environments, and the metabolic mechanisms of MPs in terrestrial organisms and humans.

2.3.2. Indirect effects of biological activities

Vegetation cover decreases the surface runoff velocity and mitigates soil erosion caused by rainfall, and both of these factors reduce horizontal soil MP migration by forming a vegetation canopy that helps minimize soil erosion [140]. Additionally, the rhizosphere of vegetation can slow surface runoff, thus reducing MP migration in surface runoff driven by precipitation [141]. Herbaceous vegetation yielded a greater interception effect than shrub vegetation, and vegetation in wetlands could remove up to 90–97% of the microfiber content from water [142].

Fig. 4 presents the mobility of MPs in soils is influenced by the soil porosity, root attachment, and root entanglement produced by plant root growth. On the one hand, plant root growth can result in a wide range of biological pore sizes, which are conducive to water infiltration [143] and promote downward MP migration in soil [11], on the other hand, the small roots of plants can capture and fix MPs in the soil, preventing their migration [144]. In particular, thin films and fibers are more likely to be captured by fine roots [98]. MPs may also come into contact with and migrate with roots, especially during apical extension and outward expansion [144].

Soil animals such as earthworms, mites, collembolans, and digging mammals facilitate MP migration by creating large pores and adhering to MPs [145, 146]. It has been shown that earthworms can transport PE particles from the surface to depths of 10-15 cm [93, 147], and Maaß et al. observed that MP migration driven by hoppers depends on the size of the organism and the MPs; notably, the larger the organism is, the higher its MP transport capacity [146].

MP migration accompanies agricultural irrigation, river dredg-

ing, farming activities, and other human activities. For example, in Japan, irrigation caused seeds and fertilizer coatings used in agricultural operations to accumulate at irrigation outlets, and this phenomenon was associated with the irrigation season [148]. Manually dredged river sediments containing high concentrations of MPs are stored in farmlands and are then dispersed to surrounding soils and surface waters under wind and hydraulic action [149]. On the surface of farmed soils, many plastic film pieces and MPs originating from other sources are present. Surface MPs migrate directly to greater soil depths [11].

Although studies have examined the influences of plants and soil organisms on MP migration, more research is needed to better understand the relevant mechanisms and to correlate them with migration efficiency. Studies on the association between vegetation coverage and migration efficiency are still scarce; for instance, both the runoff velocity and runoff kinetic energy decrease with increasing vegetation coverage [150]. Simulating the flow of MPs in watershed units is a crucial component for understanding migration. There is currently a lack of research, especially field studies, on the effects of different human activities, such as irrigation and fertilization, on the vertical migration of MPs in soil.

3. Conclusion and Future Perspectives

3.1. Conclusion

MPs can be found in many environmental media, including soil, atmospheric, and aquatic habitats. MP migration in geographical environments involves changes in spatial location, which leads to enrichment and dispersal. Therefore, it is important to understand how MPs migrate to assess the extent of MP pollution. In this review, the migration mechanisms of MPs in the terrestrial environment were thoroughly summarized. Studies have shown that MPs in the terrestrial environment can migrate through various environmental media due to physical, chemical, and biological migration.

Various natural forces, including wind, and water can cause MPs to migrate over both short and long distances. This MP migration expands the pollution range. Additionally, physical-chemical interactions, such as adsorption and degradation, between MPs and environmental materials affect MP suspension and sedimentation, making these interactions important factors in MP migration in water and soil. Biological processes can change the surrounding environment, which can impact the migration of MPs. Biological processes can cause MPs to migrate between plants and animals through the food chain.

Understanding how MPs migrate within specific terrestrial environments can aid in developing a comprehensive migration model for MPs in various environmental settings. This, in turn, can help assess the extent of MP pollution.

3.2. Future Perspectives

The following studies should be conducted in the future, and more thorough comparisons and generalizations based on existing findings are needed.

The variability of MPs in size, shape, and density makes it difficult to quantify and generalize the migration process, the characteristics of MPs can change, with residence time in the environment due to aggregation, adsorption, and degradation. As a result, the effect of microplastic characteristics on the migration process needs more quantitative research, and the changes in MP characteristics in the migration process caused by these processes need to be further studied.

In these simulations, due to the short study period in most recent studies, the influence of the environment and interactions between other particles were not considered. In the future, permanent monitoring and long-term quantitative simulation studies are needed.

MP migration has been examined across various spatial and temporal scales. It was determined that atmospheric migration and sedimentation occur at a scale of 100 km [24]. Hydraulic migration typically involves ranges of several kilometers to several hundred kilometers [9], whereas leaching-driven migration in soils occurs over a scale of only a few centimeters [72, 98]. Therefore, it is necessary to clarify the study of MP migration at the spatial and temporal scales. To analyze the migration process and dominant mechanism of MPs according to specific regions, such as river basins, and combined with the life cycle of MPs, is a worthy research direction to evaluate the level of regional microbial pollution and its ecological risk.

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Authors Contributions

H.Y. (PhD student) collected data and wrote the manuscript. W.Q.Z. (Professor) revised the manuscript. L.X.Z. (Master) made table 1. T.L. (Master student) performed visualization Fig. 1- Fig. 4. C.X.H. (Professor) revised the manuscript. Y. W. (Professor) provided financial support. T.L. (Professor) revised manuscript for language issues.

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