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SEDIMENT POLLUTION (D LAMPERT, SECTION EDITOR)

A Review of Microplastics in Aquatic Sediments: Occurrence, Fate, Transport, and Ecological Impact

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Abstract

Purpose of Review Microplastic (MP) pollution is a global concern due to its prevalence and persistence in the environment. Aquatic sediments, particularly marine sediments, are considered as the potential final sink of this pollution. This review summarizes (1) the fate and transport of microplastics (MPs) in waters and aquatic sediments and (2) the ecological impact of MPs, including the interactions between MPs and microbiome, and the effects of MPs on living organisms in aquatic sediments.

Recent Findings Characteristics of MPs, water movement, and weathering conditions determine the fate and transport of MPs. These factors influence MPs' travel and inclination to settle. The interactions of MPs and microbiome can alter bacterial communities, cause MPs' biodegradation, and facilitate biofouling that subsequently changes the fate and transport of MPs. MP presence poses exposure risks to benthic organisms through direct ingestion or trophic transfer, negatively affecting not only individual organisms but also the fauna.

Summary The destiny of MPs is affected by many factors, from MPs' characteristics to water movement and weathering. Thus, future research is warranted to develop comprehensive modeling tools that include all the key factors to better understand and predict the fate and transport of MPs in aquatic environments and sediments. The potential impact from the exposure to MPs on the ecosystem of aquatic sediments is relatively less studied. More research is needed in this area, particularly from a systematic level, to understand how different biotic and abiotic factors will interact together and what the consequential impact of these interactions on ecological and human health are.

Keywords Microplastics (MPs) · Aquatic sediments · Fate and transport · Ecological impact · Microbiome, organisms in aquatic sediments

Introduction

Global plastic production has increased exponentially since the 1950s and has raised a global concern in recent years due to the improper disposal of plastic, causing the widespread distribution of plastic debris and microplastics (MPs) in the environment [1]. MPs are plastic particles that are smaller than 5 mm. Statistical analyses have estimated that around 5.2 trillion particles are floating on the sea surface, with a total weight of 269 thousand tons, among which MPs account for up to 92.4% [2, 3]. In the environment, MPs can occur as primary or secondary MPs. Primary MPs are those produced for a variety of different applications, such as those used in cosmetic products [4], and secondary MPs are often referred to as the small pieces of the used plastics that are the result of degradation from plastic debris [3–5].

Due to their adverse effects on the ecosystem and their pervasiveness/persistence in the environment, MPs have become an increasing concern in ecological health. When plastic particles enter into the water, they can accumulate in sediments under the water body. Aquatic sediments, particularly deep marine sediments, are considered as the final sink of MP pollution. The abundance of MPs in deep marine sediment is usually higher compared to terrestrial soil, water surface, and water body [6]. Aquatic sediments are more vulnerable to MP pollution due to the accumulation of MPs in sediments.

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Furthermore, the remediation of MP pollution in aquatic sediments is exceptionally challenging. Therefore, state-of-the-art reviews of MPs in aquatic sediments are warranted given that sediments play critical roles in aquatic ecosystems. Existing reviews have provided valuable information regarding sampling/detection techniques and MP occurrence in various types of sediment. However, the fate and transport of MPs and the underlying mechanisms, as well as the consequential impacts of MP presence on the ecosystem of aquatic sediments, are less discussed. This review aims to analyze recent studies to summarize knowledge in (1) the fate and transport of MPs in aquatic environments and sediments as well as the underlying mechanisms and (2) the consequential effects of MP presence on the ecosystem of aquatic sediments, including the interactions between MPs and microbiome (microbes), and the impact of MPs on living organisms. In this review, the scope of the sediments is defined as aquatic sediments including freshwater, marine, coastal, seashore, estuary, and beach sediments.

Method—Review of Available Literature

Using the Web of Science and Google Scholar resources, the reported publications in the area of MP pollution have been searched. As seen in Fig. 1, 91% of the publications were published after 2006, with 77% of all literature being released in the last 5 years. The same search has been conducted for the topic of "MPs in sediments" with much few outcomes (Fig. 1), with the majority being published in the last 10 years. We reviewed 136 publications that contain required details within the scope of this review, including (1) occurrence of MPs in aquatic sediments, (2) fate and transport of MPs, (3) modeling MPs' distribution, (4) interactions of MPs and microbiome (microbes), and (5) effects of MP presence on living organisms.

Occurrence, Fate, and Transport of MPs in Sediments

Sources of MPs

MPs can be released from primary sources that are intentionally manufactured to be the microscopic size and released to the environment after their usages, such as resin pellets found in the spillages of plastic industries and the release of microbeads from personnel care products through wastewater discharge [3, 4]. Castañeda et al. studied MP pollution in the sediment of the St. Lawrence River (Canada) by sampling 10 freshwater sites along a 320-km section from Lake St. Francis to Québec City [7]. Various sizes (0.40–2.16 mm diameter) of polyethylene microbeads were found in St. Lawrence River sediments. The mean concentration was 13,832 (± 13,677) microbeads/m^2 with the highest density being 1.4×10^5 microbeads/m^2 [7]. The mean size of microbeads at sites receiving municipal or industrial effluent was smaller (0.70 ± 0.01 mm) than at non-effluent sites (0.98 ± 0.01 mm) (Mann–Whitney test, $p < 0.001$), suggesting different MP pollution sources [7]. MPs can also come from secondary sources, including the degradation of macro debris and torn off or dust emission during wear and tear of products made with synthetic materials such as synthetic clothing, tires, and brake pads [8]. In the aquatic environment, fisheries and aquaculture have been considered as significant sources of MP pollution [9, 10]. Plastics are widely used in fisheries and aquaculture, such as boat painting/coating, nets, floats, lines, fish boxes, and packing materials. Plastic debris from lost or abandoned fisheries and aquaculture gears are regularly found in water and on beaches and are considered as one of the primary sources of plastic pollution in the aquatic environment. Although it is hard to make a global estimate of the quantity of plastic waste from fisheries and aquaculture, Jang et al. have made the first national estimate for the Republic of Korea based on survey results and other relevant information, with an annual input of

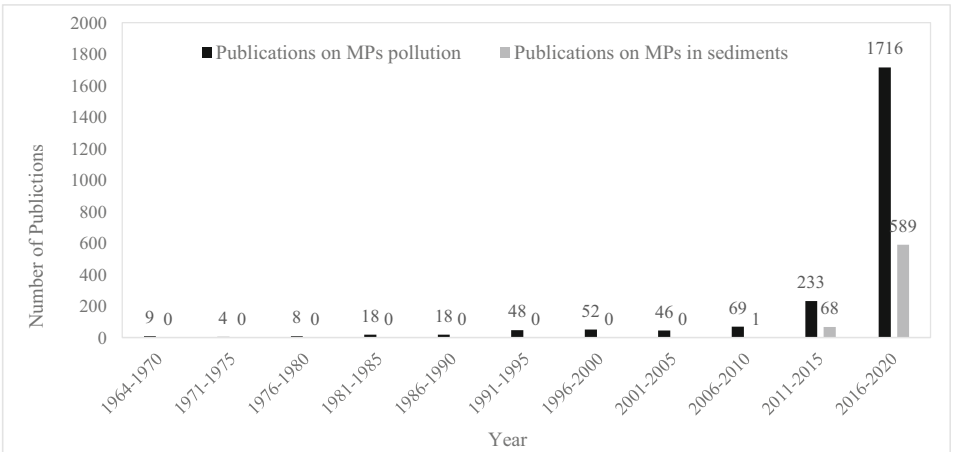

Fig. 1 Historical evolution of publications in "MP pollution" and "MPs in sediments"

44,081 tons from lost fishing gears, 2374 tons of garbage from fishing vessels, and 4382 tons of lost floats from aquaculture facilities [11].

The pollution level of MPs in the aquatic environment is not just determined by local emission rates but also affected by water current and movement [12]. For example, in studies that investigated plastic waste in the Brisbane River and the Oman Sea, 80% of marine debris is estimated to come from inland sources and be transported by rivers [13], with the rest coming from the marine industry, such as fishing, aquaculture, shipping, and private vessels [14]. Di and Wang investigated the spatial distribution of microplastics in the Three Gorges Reservoir (TGR) ecosystem located in China [15]. They reported that the MP concentration in the TGR surface waters has a tendency to increase gradually as the river flows from the suburbs (4366 ± 2855 n/m^3) to the urban areas (6201 ± 3034 n/m^3) [15]. A global analysis of plastic inputs into the oceans from rivers shows that 1.15–2.41 million tons of plastic waste reach the ocean by rivers in each year [16]. Based on the mass of the plastic waste generated in 192 coastal countries in 2010, Jambeck et al. calculated that 4.8 to 12.7 million metric tons of plastic per year reach the ocean [17]. Because MPs are translocated by water currents and their sinking is regulated by particle density and biofouling, it is hard to precisely identify the source of plastic particles in the aquatic environment [18]. Wang et al. estimated the probabilities of major sources of plastic pollution in the South China Sea and found that ropes/line/net (mainly used in fisheries) and coatings or adhesives could be the two most probable sources (83% of probability) for plastic pollution [19]. The second probable sources (67% of probability) include personnel care consumer products, textile fibers from sewage or surface runoff, spilled or recycled raw material, plastic bags and wrappers, dust from vehicle tires, packing material, and other secondary sources (decomposed from plastics) [19]. Insulation board or thermal insulation products such as styrofoam has the least source probability (17%) compared to other plastic sources [19].

MPs in Aquatic Environments and Sediments

Water movement, ocean tides, wind effects, MPs' characteristics, weathering (the rate of deterioration in the environment), and human activities would be critical in assessing the occurrence, fate, and transport of MPs and forecasting the accumulation zones [4, 20–23]. Plastic waste from consumption on upstream land would be transported by surface runoff into the river systems [24, 25]. Rivers are considered as essential pathways for transporting MPs from different areas, such as industrial coastal areas (MP hotspots), to the coast and sediment beds [26–28]. While higher amounts of MPs in inhabited areas should be assumed to be deposited directly or indirectly into the aquatic environment (discharged by sewage pipelines and terrestrial runoff), high concentrations of MPs were reported in pristine regions far from heavily populated areas [16, 21]. Therefore, it is important to understand how MPs transport from their source points to their destinies and what the underlying mechanisms are. A number of recent studies have intended to investigate the fate and transport of MPs, from beaches to isolated islands, as well as transfer from the water surface to the underlying sediments via the water column [29, 30••, 31].

The Impact of Wind and Water Movement on the Fate and Transport of MPs There is a close link between plastic distributions on the nearshore and coastal communities [25]. The coastal zone is considered as a hot spot for MP contamination due to high population and economic activity [32–34]. The transport of plastics from coastal areas to the aquatic environment is dependent on local conditions, weather patterns, wind direction/speed, water movement, and aquatic ecology [27, 29, 35, 36]. Wind pressure makes MPs traveling horizontally, so do water movements move buoyant MPs vertically through the water column. Airborne transport is particularly important for very tiny MPs that can be mobilized from untreated landfills, and affect the rate of dispersal and position of the floating waste [37, 38]. Allen et al. reported on the atmospheric deposition of MPs in a non-urban area over the 2017–2018 winter period and suggested a potential relation between wind speed and wind direction to the MP deposition [39]. Also, the beach air survey data indicated the probability of MP transmission from the ocean surface to the atmosphere, suggesting that plastic particles can leave the sea surface and enter the atmosphere [40]. In a modeling study, Critchell et al. found that the major impact on the position of debris accumulation would be the prevailing wind path [29].

Although it becomes difficult for MPs to refloat on the surface water after they reach to the sediment, a combination of wind-forced and storm-associated currents may resuspend plastics from the bottom by vertical mixing within the water column [22, 41]. The vertical distribution of MPs in the water can be controlled by wind-induced turbulence, particle density and size, bubble injection (air bubble plumes generated by breaking wave, which contains a wide range of bubble sizes and makes the bubbles injected to the water body), and Langmuir circulation [42]. In the water column, Egger et al. noted that plastic particles are mainly within the certain size range and the similar polymer composition of floating particles on the ocean floor [43]. Laboratory studies have been conducted to evaluate the settling and rise velocities of MPs as a function of density, diameter, and shapes [44–46]. For example, based on the observations of around 500 physical experiments, Waldschläger et al. reported a significant effect of the combination of the difference in density between water and particle and the particle diameter and shape on the MP rising and settling velocities [45]. The authors studied seven

types of polymer (PP, PE, PS, EPS, PVC, PET, and CoPA) with different shapes (sphere, pellet, fiber, fragment, and foam) and densities (830–1400 kg/m^3). The linear regression analysis shows that the dependence between the dimensionless particle diameter and the velocities is statistically significant, with an adjusted coefficient of determination ($r^2 = 0.832$; $p < 0.001$) [45]. The results from the multivariable linear regression analysis demonstrate that the particle shape (indicated as the Corey Shape Factor, CSF) also has a significant impact on the rising and setting velocities ($r^2 = 0.872$; $p = 0.005$), with high velocities being observed mainly at high CSF values [45]. Similarly, Khatmullina et al. measured MP settling velocities for different shapes and recorded different settling behaviors and speeds for different shapes of particles [46]. The larger the particles were, the more shape effects were observed [46]. In a field sampling study, Reisser et al. reported that vertical mixing could influence the size distribution of floating plastics on the water surface [47]. They found that plastic rising velocity, ranging from 0.001 to 0.0438 m/s, was directly proportional to plastic length [44]. Therefore, the vertical mixing process could be size-selective, which would influence the size distribution of plastics in the water column. This indicates that the absence of smaller plastics floating at the sea surface may be due in part to vertical mixing that can bring down small particles.

The Impact of MP Characteristics on the Fate and Transport of MPs The vertical distribution of plastics in the water column often relies on the density, size, and shape of the MPs [48]. Frequently detected MPs in sediments include fibers and fragments (Table 1). Different sources and shapes provide the distinction between fragments and fibers. Fibers, primarily from fabric washing, are plastic fibers (long linear shape). Fragments are considered as the breakdown of larger plastic pieces. Pohl et al. used laboratory simulations to analyze how MPs are transported and deposited by turbidity currents [30••]. Their results indicate that fragments tend to be concentrated within the base of turbidity currents, while fibers are distributed more homogeneously throughout the flow due to their large surface to volume ratio that results in the slower settlement [30••]. However, the deposit shows an opposite result, with more abundant fibers than fragments, as fibers are more likely to be trapped and dragged downward by settling sand grains [30••]. The depositional process can be determined by not only MPs' shape but also their density. Plastics with a smaller density than water are buoyant, and others with a higher density than water can settle down to the seafloor and eventually sink into sediments [3, 35]. Water movement can resuspend MPs from the benthos or transfer particles from the sediment-water interface into the water column. MPs' buoyancy enables them to distribute via water currents (water movement and ocean currents induced by wind forcing and geostrophic circulation) and probably recirculate between sediments and water [3, 20, 49]. Plastic debris with less density than water moves through surface current for very long distances [25]. Therefore, the proportion of buoyant particulates is correspondingly increased with distance from the shore [38]. On contrast, high-density MPs are more likely to be trapped in setting sand grains and settle in the sediment [35, 50]. Qualitative MP study reported the existence of polyester (PEST) (50%) and polyvinyl acetate (PVA) (25%) as the most various polymers in marine sediments [49]. These two types of plastics have relatively high density, about 1.4 g/cm^3 for PEST and 1.2 g/cm^3 for PVA [49]. Interestingly, the density of plastics can change during their residence in aquatic environments due to biofouling and degradation. In aquatic environments, MPs are easily colonized by a variety of organisms such as microbes and algae. This biofouling will increase the density, which could alter plastic buoyance. Therefore, plastics are originally buoyant could sink below the water surface and down to the sediment [3, 35]. Besseling et al. conducted a simulation analysis by using the NanoDUFLOW model and concluded that the formation of biofilms would change plastics' density and could be one of the significant mechanisms for regulating the fate and persistence of MPs [51]. On the other hand, Hidalgo-Ruz et al. found that plastics' specific density might decrease through extended exposure to the aquatic environment due to weathering [52].

The size distribution of plastic particles in the aquatic environment is impacted by a variety of factors, including biofouling and aggregation, flocculation, fragmentation, the time of residence, and transportation routes. Size and buoyancy probably determine how long MPs can float in water, thereby affecting their movement, sink rates, and tendency to settle in aquatic environments [38]. It is suggested that the fragmentation process and residence time determine the shape of plastic fragments. Sharp edges may be the result of either the recent entry into the environment or the recent break-up of larger parts, while smooth edges may represent the older fragments that have been constantly refined by certain particles or sediments [50]. As shown in Table 1, among the different types of MPs (fibers, fragments, foams, films, and spherules) reported in 45 references that were reviewed in this paper, fibrous MPs have the most occurrence in aquatic sediments, with an occurrence of 67% in the referred 43 studies. Other commonly occurring MPs include fragments (56% of occurrence) and films (19% of occurrence). Pellets, spheres, sheets, and foams are much less frequently reported in sediments (~2% of occurrence).

The Impact of Weathering on the Fate and Transport of MPs Weathering mechanisms can control the destiny of plastic waste in aquatic environments and sediments, as they are important implications for the particle's state and behavior. Larger plastic debris degrades into small fragments as a result of mechanical breakdown, photodegradation, and possibly

Table 1 The distribution of MPs in aquatic sediments

Ref.	Sediment environment	MPs present in the sediment	
		Major polymer type	Shape
[58]	Coastal	PE, PTFE, ABS, PETE, PC	Fibers, fragment, film
[59]	Coastal (wetland)	PP	Fragment
[60]	Coastal	Rayon, PP	Fiber, fragment
[61]	Coastal	PE, PP	N/A
[62]	Seashore	PE, Nylon, PET	Fiber, film
[63]	Seashore	Cellophane, PET	Fiber, fragment
[14]	Littoral (seashore)	PE, PP, Nylon	Fiber, fragment
[64]	Seashore (surface sediment)	Rayon, PP, PET	Fiber
[65]	Beach	Poly(dimer acid-co-alkyl polyamine), PP, melamine, PVF, PBR, PS, poly(perfluoroethylene oxide), polyvinyl benzoate, PVC, nylon-6, epoxy epichlorhydrin, acrylonitrile butadiene styrene	Fragment, fiber
[66]	Beach	PE, PP	Fiber, fragment
[67]	Beach	PP, PE, PVB	Fragment
[68]	Estuary	Rayon, PP, PE	Fiber
[69]	Estuary	PES, LDPE, PP	Fiber, film
[70]	Estuary	RY, PS	Fiber
[71]	Estuary	PET, PP	Fiber, fragment
[72]	Estuary	Nylon, PS, PE	Fragment, fiber
[73]	Marine	Rayon, PES	Fiber
[56]	Marine	PS, PP	N/A
[74]	Marine	Rayon, PE, PET	Fiber
[75]	Marine	PP, HDPE, LDPE	Fragment, pellet
[76]	Marine (surface sediment)	PP, PET	Fiber, film
[77]	Marine (surface sediment)	PP, PE, Nylon	Film, fiber
[78]	Marine (seabed)	PS, PMMA	Fiber, fragment
[79]	Marine beach sediments	PS, PE, PP	Fragment, fiber
[80]	River system, estuary, and lake	PET, HDPE	Foam, fiber
[81]	River (surface and deep sediment)	PE, PS	Fragment, fiber
[82]	River (river shore sediment)	PE, PP, PS	Fragment, sphere, fiber
[83]	River	PE, PA	N/A
[84]	River	PE, PA	Film, fragment
[85]	River	PE, PVC	Fragment, fiber
[86]	River	PE, PP	Fragment
[87]	River	PP	Sphere
[88]	River	PE, PP	Fiber
[89]	River	PE, PS, PP	Fragment
[90]	River	PE, PP, PS	Fragment, foam
[91]	River	PE, PP	Fiber, fragment
[92]	River	PE, PVC, PS	Fiber, film
[93]	River	PS, Nylon	Fiber, fragment
[15]	Reservoir (surface and sediment)	PS, PP, PE	Fiber, fragment, pellet
[94]	Lake	PE, PP	Sheet, fiber
[95]	Lake	PP, PE	Fiber

Table 1 (continued)

Ref.	Sediment environment	MPs present in the sediment	
		Major polymer type	Shape
[96]	Lake	LDPE	Fragment, film
[97]	Lake	PP, PE	Pellet, fragment
[98]	Lake	PP, PE	Sheet, line, fragment
[99]	Lake (nearshore, tributary, and beach)	PE, PS	Fiber, fragment

The distribution of MPs in sediments was summarized based on 43 references that contain required details of sediment environment, polymer type, and MP shape. *PP*, polypropylene; *PS*, polystyrene; *PE*, polyethylene; *HDPE*, high-density polyethylene; *LDPE*, low-density polyethylene; *PMMA*, polymethyl methacrylate; *PEST*, polyester; *PES*, polyether sulfone; *PVC*, polyvinyl chloride; *PA*, polyamide; *EPM*, poly (ethylene-propylene); *PET(or PETE)*, polyethylene terephthalate; *PTFE*, polytetrafluoroethylene; *ABS*, acrylonitrile butadiene styrene; *PBR*, polybutadiene; *PVF*, polyvinyl fluoride; *PC*, polycarbonates; *PVA*, poly (vinyl acetate); "*N/A*", not documented

microbial degradation over time when exposed to the environment [3, 25, 53]. Via iterative fragmentation processes, heat, sunlight, and well-aerated coastal environments are suitable for generating MPs [25]. Identifying the degradation processes of plastics can reveal how particles interact with the environment and how different factors affect their stability, transport, and fate [53]. It is believed that photochemical processes are the major causes of the breakdown of plastic debris [54]. Photo-oxidant disintegration of plastics is affected by a number of factors including temperature, oxygen/ozone level, and radiant energy such as UV or artificial light. Due to low temperatures, low oxygen/ozone levels, and low light in the deep aquatic environment, photo-oxidant degradation of plastics is generally ineffective in sediments [35]. In contrast, the degradation of plastics in the water surface can change significantly depending on the variation of temperature, oxygen level, and light. For instance, Kukulka et al. investigated the concentrations of MPs in the Atlantic and Pacific Oceans, and reported that the number of MPs increased during sea surface heating but declined during cooling [41].

Marine Sediments as the Ultimate Sink Dynamic coastal ecosystems and turbulent mixing of coastal sediments intensify the breakdown of plastic fragments and submerge MPs in marine sediment as the ultimate sink. Plastics located in marine sediments have usually been moved with considerable distances from their origins of pollution through water flow [55]. The abundance of MPs drops significantly with the increased depth through sediments [56]. However, MPs' longevity is high at depths due to the low oxygen concentration and degradation [26]. For instance, Barnes et al. estimated that the longevity of plastic could be hundreds or thousands of years and would probably be much longer in deep sea [26]. The quantity of MPs also varies among different grain sizes of sediment. Falahudin et al. reported that the level of MPs in a shallow and semi-enclosed bay of the Indonesia Banten Bay was affected by sediment characteristics including soil type and grain size, positively correlated with silt ($r = 0.71$, $p < 0.05$) but negatively correlated with sand ($r = -0.7$, $p < 0.05$) [57••]. However, in a study that investigated MP distribution in coastal shallow sediments in the Mediterranean Sea, the results show that there is no clear association between the quantity of MPs and sediment grain size, suggesting that other processes such as the aggregation of organic materials may play more critical roles in MPs' movement [21]. Insights into the impact of total organic carbon (TOC) on MPs transportation are of considerable significance in sedimentary environments. Areas with high concentrations of TOC could be the hotspots for MPs in aquatic sediments [57••]. High concentrations of organic matter in the water column could speed up MP aggregation and biofouling and lead to sediment deposit [53].

Ecological Impact—Interactions Between MPs and Microbiome

Biodegration of MPs in Presence of Microbes

Aquatic sediments have relatively high organic matter contents and, consequently, high levels of microbial activity. Although MPs are less susceptible to degradation than other materials, the surface of MPs can support microbial colonization and growth while presenting a carbon source to microorganisms [100]. Harrison et al. discovered bacteria found in coastal marine sediment could rapidly colonize on low-density polyethylene MPs, with evidence for the sequential formation of plastisphere-specific bacterial assemblages. Another study suggests that biofilm accumulation on PVC, PP, PE, PS, and polyurethane (PU) MPs in the Haihe Estuary of Bohai Bay, China, is not only affected by the total nitrogen and total phosphorus levels in organic matter, but

there is also a negative correlation between the biofilm growth and increasing levels of salinity [101]. This evidence indicates that freshwater sediments and higher-altitude marine sediments, where salinity is generally lower due to the decline in experienced evaporation, may be even more susceptible to bacterial growth on MPs than lower altitude marine sediment. For instance, Auta et al. isolated two bacterial isolates commonly found from mangrove sediment in Peninsular Malaysia, *Bacillus cereus* and *Bacillus gottheil*, and tested them for their biodegradability potential on various UV-treated MP polymers [102]. After 40 days of incubation, the extent of degradability was estimated by the weight loss percentage of the polymers following incubation. The *B. cereus* isolate reduced the weight of PE by 1.6%, PET by 6.6%, and PS by 7.4%. The *B. gottheil* isolate reduced the weight of PE by 6.2%, PET by 3.0%, PP by 3.6%, and PS by 5.8%.

Although MPs' biodegradation in the environment is desirable, there are some concerns regarding the effects of biodegradation on the physical and chemical properties of MPs. Because of the physical and chemical properties of MPs, multiple chemical pollutants present in the surrounding environment have been shown to adsorb to MPs, such as polycyclic aromatic hydrocarbons (PAHs), polychlorinated biphenyls (PCBs), antibiotics, organochlorine pesticides, and heavy metals [103]. For instance, in a studying measuring partition coefficients between MPs (PE, PP, and PS) and seawater for PAHs, hexachlorocyclohexanes (HCHs), and chlorinated benzenes (CBs), the results indicate that MPs pose as carriers for these organic contaminants in the marine environment [104]. As bacteria colonize on the surfaces of MPs in sediments, biofilms are often formed and cause physiochemical changes that may further enhance the sorption capacity for organic contaminants.

MPs Biofouling

Biofouling applies to the deposition of organisms on MPs in the water, which can result in increasing the density of MPs, decreasing of their buoyancy, and settling MPs in the sediment [3, 21, 42, 57••]. MPs' shapes and characteristics, biochemical conditions (e.g., temperature, photosynthetically active radiation, and availability of nutrients), and water conditions (velocity, oxygen level, light, and nutrients level) control the biofouling rate. Consequently, MPs may hold different positions at different times in the water column. The creation of biofilms often plays a significant role in determining the fate of aqueous MPs. Biofouling and microorganism adhesion increased the density of MPs, which explain why Wu et al. found MPs with low density in the sediments of Bohai Bay coast [12]. As a consequence of biofilm forming, buoyant MPs can be spread vertically through the water column and in the water surface horizontally. The formation of biofilms can also affect abiotic aging processes, for example, by slowing the degradation of floating plastics from UV radiation [105].

Biofilm growth on MPs has been correlated with changes in crystallinity, stiffness, and maximum compression. In an experimental study by McGivney et al., during a 2-week exposure to bacterioplankton assemblage from the Baltic Sea, PE MPs showed an increase in crystallinity ($X_c > 82\%$), PP MPs showed a decrease in stiffness by an average of 35 N/mm, PS MPs showed an increase in maximum compression (ε_{max}) with the exposed PS being more resistant to breaking down, and both PP and PS MPs showed significant changes in surface chemistry as indicated by the significant difference of ATR-FTIR spectra of MPs between biofilm treatments and controls [106•]. The reason for these physiochemical changes is not yet clearly known, but may be due to the biodegradation of additives present in these polymers [106•]. The potential changes caused by the degradation of additives lead to a decrease in hydrophobicity in MPs, which potentially decreases the sorption capacity of organic contaminants. Increases in negative surface charges on MPs have also been observed during this process, which may enhance the role of electrostatic interactions in sorption processes and, in contrast, increase the absorption capacity of organic contaminants in freshwater and seawater [107].

Additionally, biofouling on MPs may result in the migration and transportation of bacterial communities in aquatic ecosystems. When comparing the colonization of bacteria on MPs to the colonization of bacteria on natural substrates, alpha diversity (richness, diversity, and evenness) of bacterial assemblages are higher in natural-substrate communities [108, 109]. In a study conducted by Miao et al., freshwater bacterial communities were evaluated for 21 days in a controlled environment by incubation of biofilms on the surface of MP polymers, PE, and PP, as well as natural substrates, wood, and cobblestone [108]. Although alpha diversity was found to be lower in MP-substrate communities, higher abundances of *Pirellulaceae*, *Phycisphaerales*, *Cyclobacteriaceae*, and *Roseococcus* were found on the MP substrates. Similarly, Wu et al. analyzed bacterial communities on three substrates (MPs, surface water, and sediment) in estuarine areas (the Haihe Estuary in Bohai Bay, China) [109]. They found that the MPs could be messengers facilitating the bacterial transportation between water and sediment, enrich the particular bacteria (e.g., *Halobacteriaceae* and *Pseudoalteromonadaceae*), and weaken microbial diversity in the environment [109]. In addition, the results show that MPs carry a significantly higher abundance of potentially pathogenic bacteria (e.g., *Pseudomonas* and *Bacillus*) than the ambient environment [109]. These results indicate that the introduction of MP substrates to aquatic environments may not only alter the structure, composition, and functional properties of bacteria [108] but also lead to potential ecological risks because of the high stability, pathogenicity, and stress tolerance of the bacterial communities on MPs [109].

Finally, the co-biofouling of microorganisms and organic matters on MPs could be one of the critical factors that affect the fate and transport of MPs in aquatic environments. In addition to the chemical, mechanical, and surface charge changes of MPs, a study found the density of MPs increases in the presence of marine sediment by a combination of biofouling and organic material content [12]. The increased density can facilitate the settling of MPs; thus, marine and freshwaters with high organic matters tend to have higher concentrations of MPs in their sediments. The density change due to co-biofouling could be the cause of this migration.

Ecological Impacts—Effects of MP Presence on Living Organisms

Living organisms can ingest MPs in a variety of aquatic (e.g., marine and freshwater) and sediment environments [110, 111•, 112]. Potential risks of MP ingestion by aquatic organisms include, but are not limited to, blockages and physical abrasions [110]. These organisms may also be susceptible to varying levels of toxicity from MP additives and monomers, sorption of hydrophobic persistent organic pollutants, pathogens and microbial organisms that have colonized MPs, and heavy metals [110].

The Occurrence of MPs in Benthic Organisms

MP occurrence in sediment and aquatic organisms is affected directly by the level of MP pollution. With multiple sampling sites, this MP abundance can vary across locations [113]. For instance, across varying sample sites in the coastal environment of Suva, MP abundance in sampled species increased along with the amount of MPs in sediment samples [114]. This variance can be due to multiple reasons. In a study by Li et al., the authors noted an increase in MP ingestion by *Mytilus edulis* off the coastlines of China in areas with frequent human activities [115]. In certain studies, demersal fish species can serve as biomonitors for marine MP pollution [108, 110]. For example, a study along the Jeddah coast showed MP pollution in near shore sediments and local fish species, indicating a threat to the marine environment in the Saudi Red Sea [111•].

A study by Wang et al. found positive correlations between MP detection in benthic organisms and sediments [116•]. This study investigated sediment and organism samples collected from the South Yellow Sea; MP extraction from sediment and organism samples also indicated that MP abundances in both samples were positively correlated with water depths. The average MP abundance per water depth group varied from 1765 n/kg to 2771 ± 969 n/kg in sediment; these values were positively correlated with the average water depth, resulting in an R^2 value of 0.982. The authors of this study found that this positive correlation indicated that the deeper the seabed, the greater the MP abundance. Other studies such as one by Zhang et al. detected MPs in all surface seawater, sediment, and fish specimens sampled from artificial reefs around the Ma'an Archipelago, a national marine ranching area in China [110]. The two fish species with the highest MP abundance, *Chelidonichthys kumu* and *Muraenesox cinereus*, were found in demersal habitats and had predatory habits, indicating that MP ingestion by fish is influenced by MP pollution in sediment [110].

On the other hand, Patterson et al. reported that MP occurrence, shape, and size in Indian edible oysters (*Magallana bilineata*) are more closely related to those found in the marine surface water samples than in sediment [117]. Furthermore, a study by Bucol et al. found that semi-synthetic microfibers detected in marine subtidal sediment samples from one testing site were not the same MP type found in the guts of rabbitfish (*Siganus fuscescens*) [118]. These differences are most likely due to the transport of MPs via the food web through varying location feeding habitats [118]. Other studies have also shown possible MP transport across the benthic food web [112, 119••]. For instance, in the study by Zhang et al., the two fish species with MP presence also feed on crustaceans, which indicates MP transport via trophic transfer [110]. Similarly, Ferreira et al. observed MP occurrence in primarily benthic and demersal feeders which feed on benthic invertebrates and detritus [114]. In Corpus Christi Bay, TX, blue crabs (*Callinectes sapidus*) have been exposed to MP pollution because of their feeding patterns in dense sediment areas [120]. These studies indicate that MP occurrence in organisms can result from transport across the food web, suggesting that more comprehensive studies are needed to understand how organisms could be impacted by MPs, not only from the exposure to MPs in the environment but also from different transport pathways of MPs such as trophic transfer.

As shown in Table 2, fibers and fragments are the most common MPs detected in organisms. Plastic particles in organisms are normally found in the digestive system; however, they could be transported to organs when their size is small enough. For instance, in a sampling study by Mohsen et al., MPs can be transferred to the coelomic fluid of the sea cucumber *Apostichopus japonicus* after entering into the body [121].

Effects from MP Exposure on Organisms

The effects of MPs on the reproductive, digestive, toxicity, and development levels in aquatic organisms have been analyzed throughout various studies [112, 119, 123, 125]. These studies indicate that MPs have negative impacts on many aspects of organisms' health, including digestion, metabolism, growth and reproduction, survival, and the diversity of benthic fauna. Exposure to MPs can damage the organism's digestive

Table 2 MP occurrence in benthic organisms

Organism		MPs present in organisms			Ref.
Species	Body part	Shape/type	Size	Quantity, particles/individual or particles/kg	
Sea cucumber (*Apostichopus japonicus*)	Intestine and coelomic fluid analysis	Microfibers; cellophane and polyester	Small particles (≤ 1 mm) with maximum width of 55 μm in intestines	Avg. 10 particles per intestine; range of 0 to 19 particles animal^{-1} for coelomic fluid	[121]
Worms (*Tubifex tubifex*)	Digestion of worm tissue	Microfibers; polyester and acrylic fibers	55–4100 μm in length	Mean 129 ± 65.4 particles per g tissue	[112]
Bivalve mollusk (*Ennucula tenuis*)	Whole organism	Micro particles and irregular fragments; commercially available PE	Ranges: 4–6 μm; 20–25 μm; and 125–500 μm	2–3 particles per individual (large particles)	[122]
Bivalve mollusk (*Abra nitida*)	Whole organism	Micro particles and irregular fragments; commercially available PE.	Ranges: 4–6 μm; 20–25 μm; and 125–500 μm	1–2 particles per individual (large particles)	[122]
Chironomus tepperi	Gut content and soft tissue analysis	Regular-shaped pristine PE MPs	Ranges: 1–4, 10–27, 43–54 and 100–126 μm	MPs presence in the gut	[123]
Amphipod (*Gammarus fossarum*)	Gut and egestion analysis	Pellets (before grinding); PHB and PMMA	32–250 μm (after grinding)	4.0 ± 5.5 PHB per individual and 4.1 ± 4.9 PMMA per individual post-exposure 32 h	[124]
Nematode (*Caenorhabditis elegans*)	Intestinal analysis	PA, PE, PP, PVC, PS particles	PA, PE, PP, and/or PVC particles were ~ 70 μm in size 0.1, 1.0, and 5.0 μm PS beads	N/A	[125]
Blue crabs (*Callinectes sapidus*)	Stomach analysis	Synthetic fragments and semi-synthetic fibers	Ranging in diameter from 10 to 400 μm	0.87 items per crab	[120]
Oyster (*Ostrea denselamellosa*)	Soft tissue: flesh of bivalves	Fibers with cellulose as the dominant polymer	74–2000 μm, broken down further in Fig. 6b in article	0.31 items per g tissue	[126]
Razor clam (*Sinonovacula constricta*)	Soft tissue: flesh of bivalves	Fibers with cellulose as the dominant polymer	74–2000 μm, broken down further in Fig. 6b in article	0.21 items per g tissue.	[126]

The occurrence of MPs in benthic organisms was summarized based on field and experimental studies that contain required details of organism species (body part) and MP characteristics (shape, polymer type, size, and quantity). *PA*, polyamides; *PE*, polyethylene; *PHB*, polyhydroxybutyrate; *PMMA*, polymethylmethacrylate; *PP*, polypropylene; *PS*, polystyrene; *PVC*, polyvinyl chloride

system, causing inflammation and affecting food availability. A study in Manchester, UK, found that *Tubifex* worms sampled from the River Irwell were shown to retain MPs from bottom sediments longer than other particulate matter from the sediment matrix, resulting in negative effects such as inflammation [112]. Lei et al. observed intestinal damage on the benthic freshwater nematode *Caenorhabditis elegans* and zebrafish *Danio rerio* when exposed to 5.0 mg/m^2 MPs over 2 days in laboratory conditions [125]. Mueller et al. reported that exposed PS beads (0.1–10.0 μm) could retain in the intestinal system of the benthic nematode *Caenorhabditis elegans* and reduce food availability inside and outside of the intestine [119••]. MPs can also disturb the metabolism, undermine the growth and reproduction, and impair the survival of the organisms [119••, 122, 123, 127]. Significant dose-dependent decreases in total energy and protein content were reported in sampled bivalves *Ennucula tenuis* and *Abra nitida*, respectively, when exposed to PE particles at all concentration levels in a laboratory setup [122]. Ehlers et al. observed negative effects from MPs on larval stages, with freshwater caddisfly *Lepidostoma basale* [127]. The authors of this study suggested that MPs may transport persistent organic pollutants and emit toxic leachates, which could therein affect larval body development. Mueller et al. documented that PS beads (0.1–10 μm) could cause a 50% inhibition of reproduction for the nematode *Caenorhabditis elegans* [119••]. Ziajahromi et al. found that the survival, body length, and head capsule of benthic freshwater *Chironomus tepperi* larvae were negatively impacted by exposure to environmentally realistic concentrations of PE particles in sediment [123]. Finally, MPs not only negatively impact organisms individually but may also disturb the fauna [128]. In a study by Green, repeated exposure to high concentrations of MPs was found to reduce the occurrence of benthic fauna and thereby negatively influence macrofauna assemblages in marine environments. *Juvenile Littorina* sp. (periwinkles), *Idotea balthica* (an

isopod), and *Scrobicularia plana* (peppery furrow shell clam) were the affected benthic specimens in this study [128].

Sorption of Other Pollutants to MPs and the Consequential Impact on Living Organisms

MPs have the ability to sorb other pollutants such as heavy metals and persistent organic pollutants (POPs) from contaminated environments because of their porous surface structures and lipophilic properties [129]. A number of studies have explored the possible interactions between the abundance of MPs and possible contaminants such as heavy metals, PAHs, or total petroleum hydrocarbons (TPHs) in sediments. The results of these studies suggest that the presence of MPs is correlated to the level of these contaminants. For instance, Foshtomi et al. found significant correlations between heavy metals, intertidal sediment grain size, and MP abundance in the coastal areas of Bandar Abbas, Iran [130••]. Their results also indicate that MPs in sediment serve as a potential vector for certain PAHs and heavy metals which may result in bioaccumulation of such contaminants in the marine food web. Zuo et al. reported that sediment samples collected from mangrove wetlands in South China's Pearl River Estuary have shown a correlation between halogenated flame retardants (HFRs) and MP abundance, indicating that certain HFRs may have the same pollution source as MPs [131]. The presence of MP-sorbed phthalate esters (PAEs) has also been found in littoral sandflat sediments of the Gulf of Guinea [132], indicating medium to high biological risks from certain PAEs to marine organisms. One study by Zhang, Liu et al. found a significant correlation between PCB concentrations (sampled from pore water of deep-sea sediments) and MP distribution in the western Pacific Ocean [133]. In another study by Fraser et al., MPs and PCBs were found in all sediment samples from varying sites along the Qiantang River and Hangzhou Bay, China [134]. Authors of the study found correlations between microbeads and films MP types and certain PCB congeners. These correlations suggested a high composite toxicology of MPs through PCB sorption, indicating potential harm to aquatic organisms [134].

Studies have also been conducted to investigate the effects of sorption of pollutants by MPs on organisms. Gomiero et al. examined the effects of sorption of pollutants by PVC MPs through exposure to benthic organism *Hediste diversicolor*, an annelid worm [135]. Results from this exposure study indicated that PVC particles adsorbed B[a]P in a time and dose-dependent manner. PVC particles contaminated with B[a]P increased the rate of bioaccumulation in *H. diversicolor*, and negatively affected cellular functioning and genotoxicity in coelomocytes with permanent oxidative stress effects in tissue [135]. Contrastingly, in a study by Besseling et al. in which PE MP particles were added to sample sediment to study PCB uptake by *Arenicola marina*, results showed that PE was not a measurable vector for the hydrophobic contaminant PCB [136]. The authors of this study also indicated that further biodynamic model analysis showed marginal effects from PE ingestion on bioaccumulation.

Conclusions

This review summarizes (1) the fate and transport of MPs in waters and aquatic sediments and (2) the ecological impact of MPs, including the interactions between MPs and microbiome, and the effects of MPs on living organisms. The fate and transport of MPs in aquatic sediments are determined by many factors, including characteristics of MPs, water movement, and weathering. These factors decide how long MPs can float and thus influence their travel, settling levels, and inclination to settle. Thus, future research is warranted to develop comprehensive modeling tools that include all the key factors to better understand and predict the fate and transport of MPs in waters and aquatic sediments. This article also reviewed potential impact of MP exposure on the ecosystem of aquatic sediments. The interactions of MPs and microbiome can alter bacterial communities, cause MPs' biodegradation, and facilitate biofouling that subsequently changes the fate and transport of MPs in waters and aquatic sediments. In addition, MP presence poses exposure risks to benthic organisms through direct ingestion or trophic transfer, which not only negatively impact organisms individually but may also disturb the fauna in aquatic sediments. Future research in this area should focus on developing systematic approaches to understand how different biotic and abiotic factors will interact each other and what the consequential impact of these interactions on ecological and human health are.

Compliance with Ethical Standards

Conflict of Interest The authors declare that they have no conflicts of interest.

Human and Animal Rights and Informed Consent This article does not contain any studies with human or animal subjects performed by any of the authors.

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- •• Of major importance

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