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Review

Environmental occurrences, fate, and impacts of microplastics

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ABSTRACT

Microplastics (MPs) are small plastic pieces with size less than 5 mm that have entered and polluted the environment. While many investigations including several critical reviews on MPs in the environment have been conducted, most of them are focused on their occurrences in marine environment. Current understanding on the occurrences, behaviors, and impacts of MPs in the terrestrial environment is far from complete. A systematic review of the literature was thus conducted to promote the research on MPs in the environment. This work is designed to provide a comprehensive overview that summarizes current knowledge and research findings on environmental occurrences, fate and transport, and impacts of MPs. In addition to discussing the occurrences, characteristics, and sources of MPs in the ocean, freshwater, sediments, soils, and atmosphere, the review also summarizes both the experimental and modeling data of the environmental fate and transport of MPs. Research findings on the toxic effects, bioaccumulation, and bioavailability of MPs in the environment are also covered in this critical review. Future perspectives are discussed as well.

1. Introduction

Since plastic products started to reach the market at large scale in the 1950s, the global production of plastic dramatically increased from 0.5 million tons per year in 1960 to 348 million tons in 2017 (Barnes et al., 2009; PlasticsEurope, 2018). Because of its slow degradation, plastic is easily accumulated in the environment from various sources (Barnes et al., 2009). Plastic pollution in terrestrial and marine environments has been long and widely reported (Carpenter et al., 1972; Colton, 1974; Gregory, 1978) and thus attracted increasing public attention, especially with respect to the potential risks of microscopic resin pellets. For example, Gregory (1978) reported that over 1000 tons of resin pellets were accumulated on New Zealand beaches due to accidental spillage during transport and handling from plastic industry, causing serious contamination issues to the marine and coastal ecosystems (Mato et al., 2001). With strict laws and regulations to ban marine debris dumping and strengthening of raw material recycling management in plastic industries in 1990, the quantity of plastic debris entering terrestrial and marine environments has been stabilized (Barnes et al., 2009) and even reduced in terms of plastic pellets (Law et al., 2010). However, environmental occurrences of large quantities of microplastics (MPs) have been observed frequently in recent years due to the breakdown of existing plastic waste in the environment (Moore et al., 2001; Thompson et al., 2004). MPs are generally defined as plastic debris with the diameter or length of less than 0.5 cm (Arthur et al., 2009; Hidalgo-Ruz et al., 2012; Ryan et al., 2009).

Based on their sizes, plastic debris in the environment can be divided into four categories (Fig. 1): MPs (< 0.5 cm), mesoplastics (0.5–5 cm), macroplastics (5–50 cm), and megaplastics (> 50 cm) (Lebreton et al., 2018). MPs can be classified into primary and secondary MPs based on their sources. Primary MPs in the environment are mainly from the direct release of MP-containing products such as preproduction MP pellets that are used to manufacture plastic products and plastic microbeads as additive in personal care products (e.g., hand and facial cleansers, toothpaste, and cosmetics) (Barnes et al., 2009; Fendall and Sewell, 2009; Gregory, 1978). Secondary MPs are fragments of plastics derived from the degradation of larger plastic products (e.g. rope, packaging, and clothing) (Barnes et al., 2009; Browne et al., 2011;

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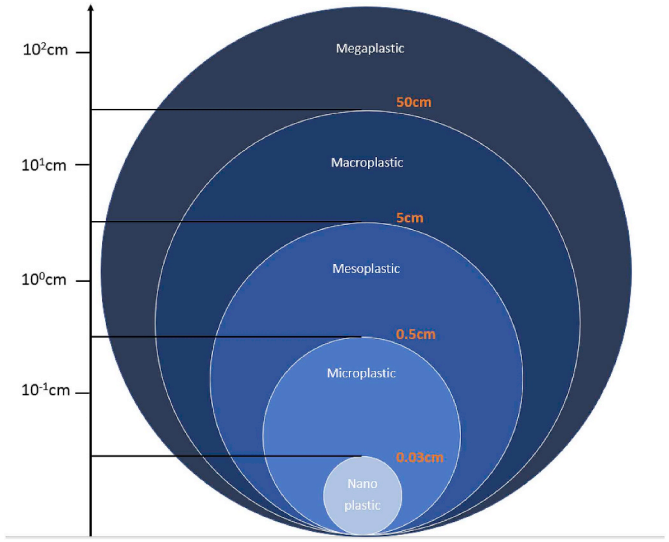

Fig. 1. Class of plastic based on size: nanoplastic (< 0.03 cm), microplastic (0.05–0.5 cm), mesoplastic (0.5–5 cm), macroplastic (5–50 cm) and megaplastic (> 50 cm) (data from reference (Andrady, 2011; Eriksen et al., 2014; Koelmans et al., 2017; Lebreton et al., 2018)).

Cole et al., 2011; Thompson et al., 2004). Researchers have also described MPs according to their shapes as microbeads, nurdles, fibers, foam, and fragments (Fig. 2) (Hidalgo-Ruz et al., 2012). Moreover, MPs have been further classified into large MPs (1–5 mm) (Horton et al., 2017b), small MPs (0.3–1 mm) (Claessens et al., 2011), and nanoplastics (< 0.3 mm) (Andrady, 2011) based on the characteristic of continuous breakdown.

Because of their persistence in the nature and potential negative effects on waterbody, wildlife, ecosystem, and human health, MP pollution has attracted much research attention recently (Gasperi et al., 2018; Ivleva et al., 2017; Lebreton et al., 2017). Several critical reviews on MPs in the environment have been published, however, most of them are focused on the effects of MPs on the marine environment (Andrady, 2011; Cole et al., 2011; Ivar do Sul and Costa, 2014; Sharma and Chatterjee, 2017; Wright et al., 2013). MPs may be generated from various sources, interact with multiple environmental media (e.g., freshwater, marine water, groundwater, sediments, and soils), and have various transport and transformation pathways (Fig. 3). In addition to the marine environment, MPs have been frequently detected in freshwater (Hurley et al., 2018; Klein et al., 2015; Lechner et al., 2014), soils (Maaβ et al., 2017; Nizzetto et al., 2016) and atmosphere (Dris et al., 2017; Gasperi et al., 2018). Nevertheless, relatively few studies have summarized the research progresses on MPs in freshwater and sediments (Blettler et al., 2018; Burns and Boxall, 2018; Horton et al., 2017b; Ivleva et al., 2017; Li et al., 2018; Rezania et al., 2018; Van Cauwenberghe et al., 2015). Furthermore, the transport behaviors of MPs in hydrological pathways have been largely overlooked in previous reviews. With the rapid expanding of academic results and emerging of new directions, there is a crucial need for another critical review on MPs in the environment.

The overarching goal of this work is to provide a comprehensive review to summarize current knowledge and research findings on occurrences, fate and transport, and impacts of MPs in the environment. Specifically, this critical review assembles and summarizes current research findings on MPs with following focus: 1) occurrences of MPs in various environments including the ocean, freshwater, sediments, soils, and atmosphere; 2) fate and transport of MPs in the environment; and

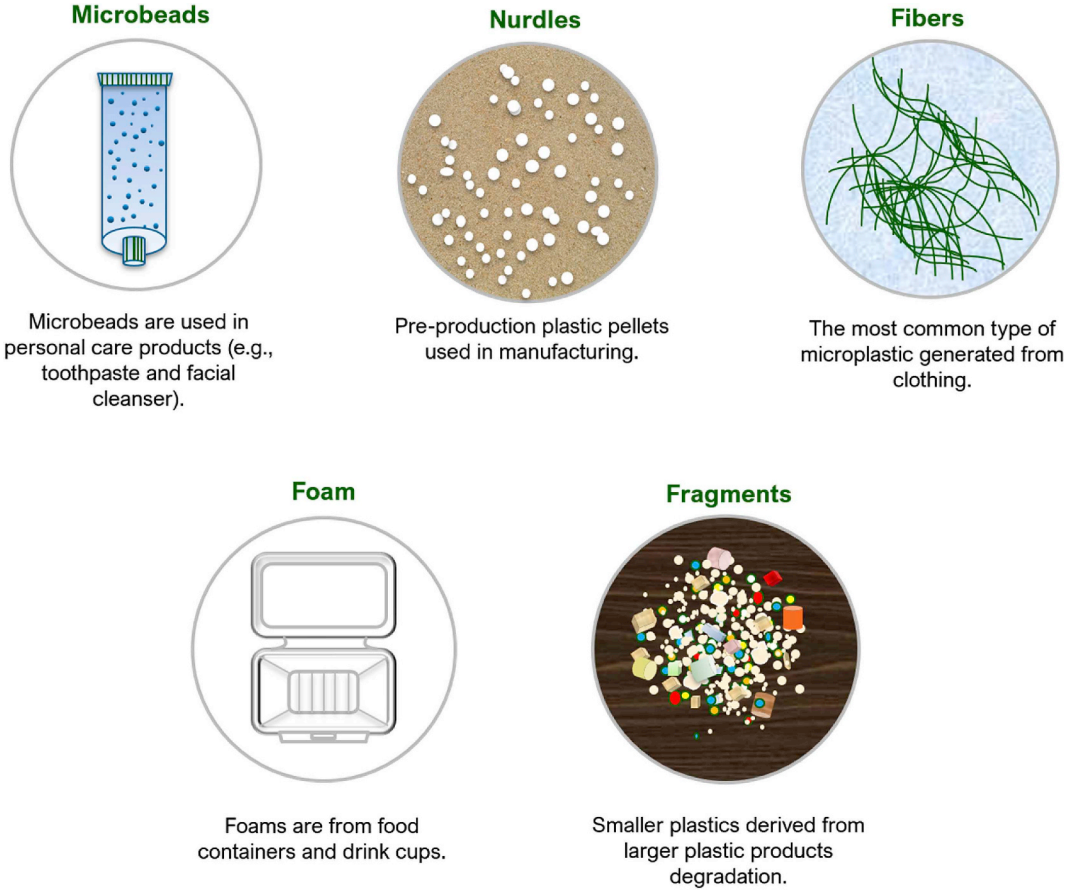

Fig. 2. Categories and sources of MPs in the environment.

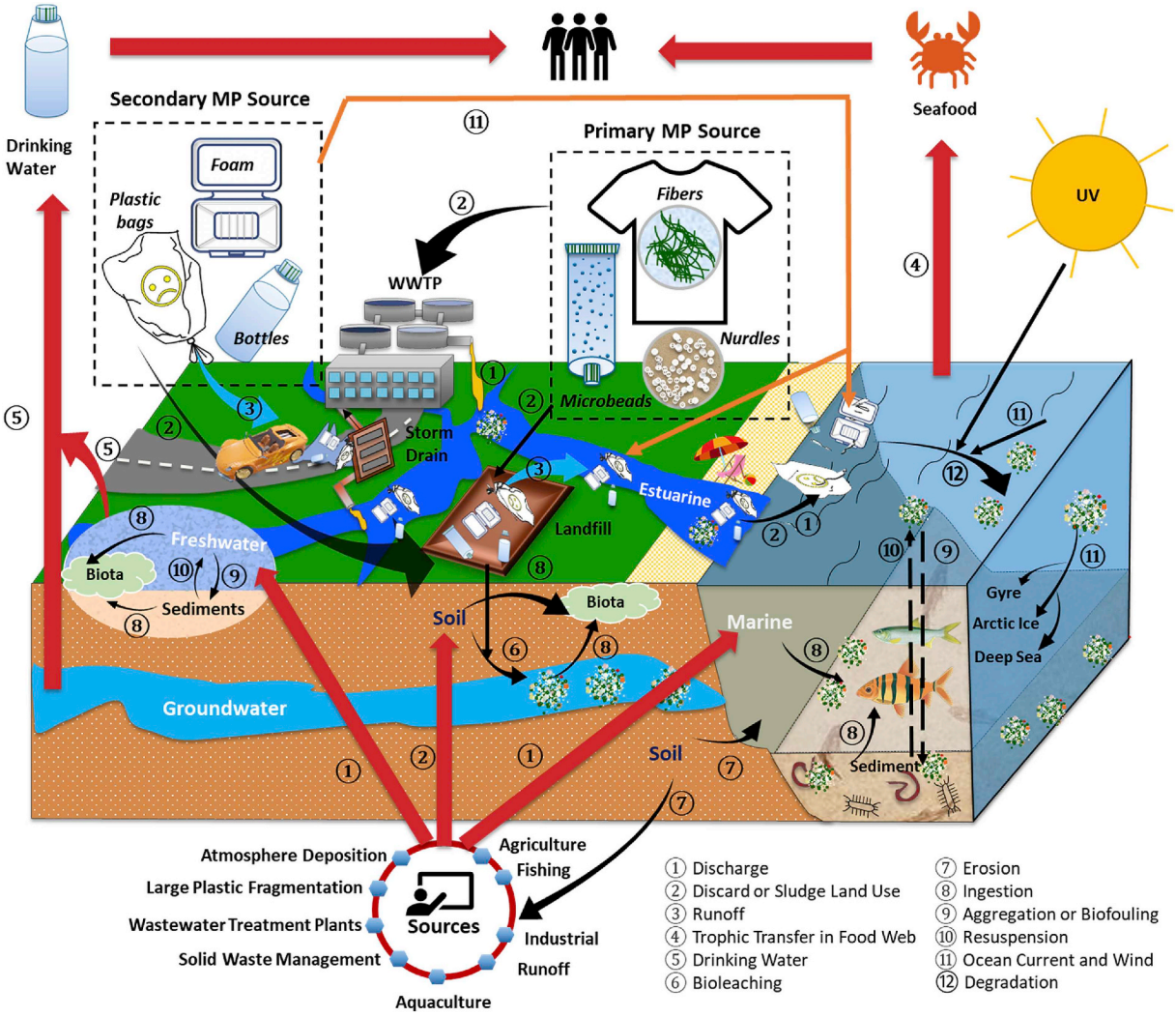

Fig. 3. Sources, transport, accumulations, and fate of MPs in the environment.

3) toxicity and bioaccumulation of MPs in the environment. Perspectives on future research direction of MPs in the environment are also discussed.

2. MPs in the ocean

2.1. Occurrences

In the ocean, the accumulation of plastics has been reported all over the world even in abyssal and polar regions (Barnes et al., 2009).

Table 1
Specific densities of different polymer types (Hidalgo-Ruz et al., 2012).

Type	Density (g/cm^3)
Polyethylene (PE)	0.917–0.965
Polypropylene (PP)	0.9–0.91
Polystyrene (PS)	1.04–1.1
Polyamide (nylon)	1.02–1.05
Polyester	1.24–2.3
Acrylic	1.09–1.20
Polyoximethylene	1.41–1.61
Polyvinyl alcohol	1.19–1.31
Polyvinyl chloride (PVC)	1.16–1.58

Because of their relatively low density (Table 1), MPs in the ocean are often floatable, leading to their pervasive occurrences in the marine environment (Table 2). They are now widespread in world oceans, from the Atlantic to the Pacific Ocean in terms of the open ocean and from the Caribbean Sea to the Mediterranean Sea in terms of the enclosed or semi-enclosed ocean (Cózar et al., 2014; Law et al., 2010; Ory et al., 2018; Welden and Lusher, 2017). Lusher et al. (2014) reported that 94% of the plastic samples in the Northeast Atlantic are MPs and 89% of the 2315 particles collected are less than 5 mm in length. The frequency of MP occurrence is 96% in the Pacific (Moore et al., 2001) and 100% in the Mediterranean surface waters (Cózar et al., 2014; Suaria et al., 2016; Vianello et al., 2013). Cózar et al. (2014) showed a worldwide distribution of MPs in the open ocean, and indicated that the frequency of occurrence of MPs in the surface samples is considerably high (88%). MPs have also been found in ice cores in the remote Polar areas of the Arctic Ocean (Obbard et al., 2014) with a frequency of occurrence of 93%–95% in the Arctic waterbodies (Lusher et al., 2015).

Since smaller MP particles have lower rise velocities (Reisser et al., 2015), their distribution in the sea water is different from the large ones with the same polymer compositions, which are often on the surface of ocean water. The larger specific surface area of smaller MPs can promote their interactions with phytoplankton, organic debris, clays, and other particles, causing an increase of their densities (Claessens et al., 2013). For example, in the North Atlantic Gyre, MPs with size of 0.5–1.0 mm are more plentiful in subsurface water than surface water

Table 2
Summary of selected microplastic environmental sampling studies of freshwater and marine system.

Water body	Location	Sample type	Mesh size	Occurrence frequency	Concentration (counts or weight)	Particle size (diameter or length)	MP type	Poly type	Reference
Lake	Laurentian Great Lakes (USA)	Surface waters	333 μm	95.2%	Mean: 43,157/km^2; Max: 466,305/km^2; Min: 0/km^2	0.355–0.999 mm (81%) 1.000–4.749 mm (17%) > 4.75 mm (2%)	Microbeads	PE	Eriksen et al. (2013)
Lake	Lake Hovsgol (Mongolia)	pelagic waters	333 μm	100%	Mean: 20,264/km^2; Max: 44,435/km^2; Min:997/km^2	0.355–0.999 mm (41%) 1.00–4.749 mm (40%) > 4.75 mm (19%)	Fragments (40%) Films (38%) Line/fiber (20%) Microbeads (0%) Pellet (1%)	–	Free et al. (2014)
Lake	6 Lakes (Swiss)	Surface water	300 μm	100%	Mean: 91,000/km^2 (26,000 mg/km^2)	< 1 mm–85%	Fragments (61%) Fibers (10%) Pellets (1%)	FT-IR:62% PE, 15% PP,12% PS; PE Films	Faure et al. (2015)
		Beach sediment	300 μm	100%	Mean: 1300/m^2; Max: 7200/m^2; Min: 20/m^2	–	Foam (46%) Fragments (27%) Fibers (19%)	PP fragment PS foams	
		Fish		7.5%	31 fibers/fish; 1 fragment/fish	–	Fragments and fibers		
		bird		88.9%	4.3 per bird	–	Microbeads and fibers		
Lake	Taihu Lake (China)	Surface water	333 μm	–	Max: 6,800,000/km^2 (3.4/L); Min: 10,000/km^2 (25.8/L)	0.333–1 mm	Fiber > 60%	Cellophane, PET, PS, PP	Su et al. (2016)
		sediment	1 mm	–	Max: 234.6/kg dry weight (dw); Min: 11/kg dw	0.1–1 mm	Fibers and fragments		
		Asian clams		–	Max: 12.5/g wet weight (ww); Min: 0.2/gww	0.1–1 mm	Fibers > 70%		
River	Urban rivers (USA)	Surface, subsurface, bottom water	333–800 μm	–	Max: 12,932/m^3; Min: 10/m^3	1.0–4.75 mm	PS Foam (82.7%) Pellets (10.6%) Fragments (6.0%) Lines (0.2%)	–	Moore et al. (2011)
River	Urban river (USA)	Surface waters	333 μm	100%	17.93/m^3 (6,698,264/km^2) (WWTP downstream); 1.94/m^3 (730,341/km^2) (upstream)	0.333–2 mm	Fibers (59.0%–62.4%) Fragments (37.1%–37.6%) Pellets (0–2.51%) Foam (0–1.39%)	–	McCormick et al. (2014)
River	Rhine river (Germany)	Surface waters	300 μm	100%	Mean: 892,777/km^2; Max: 3,900,000/km^2	300 μm − 5 mm	Spherules (58.4%) (70% cross-linked PS & 15% PP) Fragments (37.5%) Fibers (2.5%)	29.7% PS, 16.9% PP, 13.6% other types, 9.3% acrylate, 5.1% polyester, 1.7% PVC	Mani et al. (2015)
River	Danube River (Austria)	Surface water	500 μm		Mean: 0.3168/m^3 0.28 fish larvae/m^3	0.5–50 mm	79.4% pellets/flakes/spherules		Lechner et al. (2014)
River	Yangtze River (China)	Surface water	112 μm	100%	Mean: 8,465,600/km^2; Max: 13,617,500/km^2; Min: 3,407,700/km^2	0.5–1.6 mm (30–57%)	Sheet, line, fragment; no microbeads	42.1%-63.2 PP, 36.8%–57.1% PE, 0–12.7% PS	Zhang et al. (2015)
River	Seine River (France)	Surface water	80–330 μm	–	3-106/m^3 (plankton net) 0.28–0.47/m^3 (manta trawl)	< 1 mm plankton net (52%) 1–5 mm manta trawl (75%)	Fibers and beads (plankton net) Fibers, fragments, and spherules (manta trawl)	–	Dris et al. (2015)
Lake	29 Great Lakes tributaries (USA)	Surface water	333 μm	100%	Max: 32/m^3; Min: 0.05/m^3	0.355–0.99 mm (72%) 1–4.75 mm (26%) > 4.75 mm (2%)	Fibers/lines (71%) Fragments (17%) Pellets/beads (2%)	–	Baldwin et al. (2016)

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Table 2 (*continued*)

Water body	Location	Sample type	Mesh size	Occurrence frequency	Concentration (counts or weight)	Particle size (diameter or length)	MP type	Poly type	Reference
River	4 Chesapeake Bay Estuarine Rivers (USA)	Surface water	330 μm	98%	Max: 297,927/km^2	0.3–2.0 mm	Fragments, sheets and fibers	PE	Yonkos et al. (2014)
River	Rhine-Main area (Germany)	Shore sediment	63–200 μm	100%	Mean: ~4000/kg Mass mean: 1 g/kg	630–5000 μm (most abundance size) 200–630 μm 63–200 μm	Pellets	75% PE/PP/PS	Klein et al. (2015)
River	St. Lawrence River (Canada)	Basin sediment	500 μm	80%	Mean: 13,759/m^2; Max: 136,926/m^2	0.40–2.16 mm (Mean: 0.76) 0.70 mm (effluent) 0.98 mm (non-effluent)	Microbeads	PE	Castañeda et al. (2014)
River	Thames River (UK)	Basin sediment		100%	Max: 660/kg (downstream of urban runoff)	1–2 mm (55–63%) 2–4 mm (37–45%)	Fragments (urban runoff downstream, 91%) Fibers (other sites, > 67%)	PP, PET, polyarylsulphone thermoplastic	Horton et al. (2017a)
Lake	Garda Lake (Italy)	Beach sediment		–	1108/m^2 (north shore); 108/m^2 (south shore)	< 5 mm	Fragments and fibers	45.6% PS, 43.1% PE, 9.8% PP	Imhof et al. (2013)
Marine	Beach Continental shelf (UK)	Sediment Sediment		– –	0.4/50 mL 5.6/50 mL	~20 μm	Fibers	9 polymers	Thompson et al. (2004)
Marine	Beach Continental shelf (Belgium)	Sediment Sediment	38 μm 38 μm	100% 100%	92.8/kg 97.2/kg	38 μm-1 mm	Fibers (59%) Granules (25%)	PP, nylon, PVA	Claessens et al. (2011)
Marine	Irish Continental shelf (Atlantic)	Sediment Bottom water	250 μm	90.9%	97% water-sediment interface + top 0.5 cm	250 μm- 5 mm	Fibers (85%) Fragments (15%)	23% nylon, 11% polyester, 3% PP, 2% acrylic	Martin et al. (2017)
Marine	Italy Continental shelf (Mediterranean)	Sediment	32 μm	100%	Max: 2175/kg; Min: 672/kg	< 1 mm (S-MPPs) 93% 30–500 μm	Fragment (87%) Fibers (10%) Films (2%) Pellets/granules (1%)	82% PE and PP	Vianello et al. (2013)
Marine	Spain Costal shallow waters (Mediterranean)	Sediment	63–2000 μm	–	Max: 0.90/g (MPAs)	Most: 1–2 mm, 0.5–1 mm	Fragments (MPAs, > 60%) Filaments (near the Populated areas, > 60%)	–	Alomar et al. (2016)
Marine	Canada Beach (Atlantic)	Sediment mussels	63–500 μm	–	Mean: 20–80/10 g; 375/5 farmed-mussels, 170/5 wild mussels	–	Fibers	–	Mathalon and Hill (2014)
Marine	Deep sea (1000–3500 m) of Atlantic, Mediterranean Sea and Indian Ocean	Sediment	> 32 μm	100%	Mean: 13.4/50 mL; Max: 40/50 mL; Min: 1.4/50 mL 4 billion fibers/km^2 (Indian Ocean)	2–3 mm in length < 0.1 mm in diameter	100% Microfibers	56.9% Rayon 53.4% polyester, 34.1% nylon, 12.4% acrylic	Woodall et al. (2014)
Marine	Deep sea (1176–4843 m) of Atlantic Ocean and Mediterranean Sea	Sediment	35–1000 μm	75%	Mean: 0.5/25 cm^2; Max: 1/25 cm^2	75–161 μm	–	Non-natural organic pigments	Van Cauwenberghe et al. (2013)
Marine	Deep-sea of Mid-Atlantic and Indian Ocean	Organisms		66.7%	Were found inside oral or stomach area	–	100% Fibers	Modified acrylic, PP, polyester, viscose, acrylic	Taylor et al. (2016)
Marine	Polar waters (Arctic)	Surface (16 cm) subsurface (6 m) water	333 μm	93%–95%	Surface Mean: 0.34/m^3; Max: 1.31/m^3; Subsurface Mean: 2.86/m^3; Max: 11.5/m^3	0.25–7.71 mm (mean: 1.93)	Fibers (95%) Fragments (4.9%)	30% Rayon 15% polyester 15% nylon (polyamide) 5% PE	Lusher et al. (2015)
Marine	Polar waters (Arctic)	Sea ice		–	38-234/m^3 ice	2 mm (fibers) 0.02 mm (nodules) < 0.2 mm (chips)	Fibers Blue nodules Orange chips	54% Rayon, 21% polyester, 16% nylon 3% PP, 2% PS, 2% PE	Obbard et al. (2014)

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Table 2 (*continued*)

Water body	Location	Sample type	Mesh size	Occurrence frequency	Concentration (counts or weight)	Particle size (diameter or length)	MP type	Poly type	Reference
Marine	Atlantic	Subsurface water	250 μm	94%	Mean: 2.46 ± 2.43/m^3	89% < 5 mm 0.2–43.2 mm most: 1.25–2.5 mm	Fibers (95.9%) Fragments (3.86%)	Polyester and Nylon	Lusher et al. (2014)
Marine	NE Pacific Ocean and coastal British Columbia	Subsurface (4.5 m)	> 62.5 μm	–	Max: 9180/m^3; Min: 8/m^3	0.1–1 mm (79.2%) < 0.1 mm (5.92 %) > 1 mm (14.9%)	Fibers (75%)	–	Desforges et al. (2014)
Marine	Southern Portuguese shelf (Atlantic)	Sediment	1–500 μm	56%	–	–	Microfibers (80.6%) Fragments (19.4%)	Rayon fibers and PP fragments	Frias et al. (2016)
Marine	Italy (Mediterranean)	Surface water	200 μm	100%	1.25/m^2	< 300 μm (26%) < 500 μm (51%) > 5 mm (1.4%)	Fragments (93.2%) Pellets (2.2%)	52% PE 16% PP (16 polymer types)	Suaria et al. (2016)
Marine	NW (Mediterranean)	Surface water organisms <	0.333 mm	90%	Mean: 0.116/m^2 (0.202 mg/m^2); Max: 0.892/m^2 MPs/mesozooplankton weigh ratio = 0.5	0.3–5 mm	Filaments PS Films	–	Collignon et al. (2012)
Marine	Subtropical Ocean Gyres (Pacific)	Surface waters	0.5 mm	99.9%	Mean: 678,000/km^2; Max: 11,054,595/km^2; Min: 20,108/km^2	0.5–5 mm (94%)	–	–	Lebreton et al. (2018)
Marine	Subtropical Ocean Gyres (Mediterranean)	Surface water	200 μm	100%	Mean: 0.25/m^2 (243,853/km^2); Mean mass: 243 g/km^2 accumulation load: 1,000–3,000 tons	83% < 5 mm	Fragments (87.7%) Films (5.9%) Foam & fishing thread (both 2.3%) Pellet/granule (PCPs, 1.8%)	–	Cózar et al. (2015)
Marine	Open-ocean (global)	Surface water	0.2–1 mm	88%	6.6–35.2 thousand tons	1–5 mm	Fragments Resin pellets (2%)	PE	Cózar et al. (2014)
Marine	Open-ocean and Mediterranean Sea (global)	Surface waters	0.33 mm	92.3%	Mean: > 5.25*10^{12} (268,940 tons), 890,000/km^2 (Mediterranean Sea)	< 4.75 mm (92.4%) 0.33–1.00 mm (34.9%) 1.01–4.75 mm (57.5%)	Fragments	–	Eriksen et al. (2014)
Marine	Global ocean	Surface waters	0.333 or 0.335 mm	–	15-51*10^{12} (93–236 thousand tons)			–	Van Sebille et al. (2015)
Marine	Global ocean	Surface waters	0.3 mm	–	309 thousand tons	Macroplastic (~83.7%) MPs (13.8%) Nanoplastic (2.5%)	-	–	Koelmans et al. (2017)

(Reisser et al., 2015). Moreover, the bottom water is contaminated by MPs almost everywhere along the western Irish continental shelf and 66% of recovered microplastics are found at the water-sediment interface (Martin et al., 2017). Therefore, MPs are widespread throughout the water column and sediment (Alomar et al., 2016; Martin et al., 2017). Unfortunately, the distribution behaviors and mechanisms of MPs are still unclear, further investigations are needed to clarify the vertical profiles of MPs of various size ranges for better predicting and monitoring of MP occurrence in the marine system (Takada and Tanaka, 2016).

In the past few decades, there is an increasing tendency of global MP abundance and dispersion (Barnes et al., 2009; Lebreton et al., 2018; Pham et al., 2014). Concentration of MPs in water has been reported from a low (e.g., 0.34 particles/m^3 (Lusher et al., 2015)) to very high (e.g., 102,000 particles/m^3 (Norén and Naustvoll, 2010)) level. Though MP concentration ranges widely among the open sea (Cózar et al., 2014), their abundance is more evident in the areas characterized by convergence currents, where litter is accumulated (Moore et al., 2001). Recently, both field surveys and models have demonstrated that floating MPs exist among five large accumulation zones in the open sea corresponding to convergence zones (e.g. subtropical gyres) (Eriksen et al., 2013; Goldstein et al., 2012; Law and Thompson, 2014), and their distribution pattern matches those data predicted from ocean surface circulation models (Law et al., 2010; Maximenko et al., 2012). These five accumulation zones are distributed northwards and southwards of the equator, including the North Pacific subtropical gyre, the North Atlantic subtropical gyre, the South Pacific subtropical gyre, the South Atlantic subtropical gyre, and the Indian Ocean. High concentrations of MPs (up to the order of kilograms per km^2) are found in those regions (Law and Thompson, 2014), while open-ocean concentrations only reach a few grams per km^2 occasionally (Goldstein et al., 2012; Law and Thompson, 2014; Lebreton et al., 2018).

Given what have been mention above, nowadays, there is a heavy MP pollution in the marine environment and MPs are the most abundant components in marine plastic debris. The average size of MPs in the environment seems to be reducing (Koelmans et al., 2017); however, this may underestimate real accumulation of MPs if the data are only from ocean surface water. Additionally, the MP distribution is easily affected by natural condition so that it is not well-distributed, and thus the spatial distribution of MPs is irregular. With the shortage of detailed monitoring points and sections, and standardized and accurate measurements, some academic results are not consistent, causing trouble to account global marine pollution of MPs. This is an austere challenge of understanding the MP pollution in the marine environment.

2.2. Characteristics

Recent studies have indicated that fibers and fragments are the most common types of MPs in the ocean and they may be up to 80% of total, much more than that of pellets/granules (Desforges et al., 2014; Lusher et al., 2014). For example, a research of the Mediterranean Sea showed that 87.7%–93.2% of MPs are fragments in surface water, only 2% of which are pellets/granules (Cózar et al., 2015). The percentage of fragments and microfibers varies spatially. In the Arctic, fibers are the most abundant (95%), followed by fragments (4.9%) (Lusher et al., 2015). However, in the Mediterranean Sea, the percentage of fragments can be up to 93.2% over all (Suaria et al., 2016).

It is important to identify the sources of MPs by identifying polymer types (Takada and Tanaka, 2016). Polyethylene (PE), polypropylene (PP) and polystyrene (PS) are the most commonly reported plastic debris in the surface water worldwide (Frias et al., 2014; Suaria et al., 2016). Hidalgo-Ruz et al. (2012) reviewed the types of MPs sampled at marine/sediments and found the dominance of PE, PP, and PS. The result approximately correlates with plastic production worldwide with 62% of global demands coming from PE and PP (Andrady, 2015). In addition, these polymers have lower density than seawater (Table 1), it is thus not surprising that they consistently predominate in surface water (Table 2). Less frequently detected polymers including polyethylene terephthalate (PET), polyisoprene (synthetic rubber), poly (vinyl stearate) (PVS), ethylene-vinyl acetate (EVA), polyepoxide (epoxy resin), paraffin wax, and polycaprolactone (a biodegradable polyester) have also been reported floating in off-shore waters (Suaria et al., 2016). Moreover, rayon (a semi-synthetic cellulosic material used in cigarette filters), personal hygiene products, and clothing can enter the ocean through sewage (Barnes et al., 2009). These man-made semisynthetic materials are classified as synthetic MPs and may make up a significant proportion of MPs in the marine environment (Lusher et al., 2013; Obbard et al., 2014). The variation of MP types in the ocean may provide useful information for identifying the sources of MPs and thus to improve current management practices of plastic waste (Rochman et al., 2015).

2.3. Sources and pathways

The sources of plastics in the ocean include fragmentation of mega- and macro-plastic items entering from rivers, runoff, tides, winds, and catastrophic events, together with at-sea sources, including lost cargo (Thompson et al., 2004) and fishing (Bell et al., 2017; Watson et al., 2013) and aquaculture gear (Law and Thompson, 2014). Lebreton et al. (2018) found that most collected plastics, made of PE and PP rigid plastics and bundled fishing nets and ropes, are from marine based sources. About 80% of marine plastic debris originate from inland sources and are transmitted by rivers to the oceans (Kershaw and Rochman, 2015; Mani et al., 2015). “River to the ocean” is one of the most important pathways for the major plastic reservoirs (Lebreton et al., 2017). One study estimated that the Danube River released 530–1500 tons of plastic into the Black Sea annually (Lechner et al., 2014). A global model of plastic inputs from rivers into oceans indicates that 1.15–2.41 million tons of plastic waste enter the ocean through rivers annually (Lebreton et al., 2017). The information mentioned above can refine the understanding of the sources and pathways of MP contamination in the ocean.

The direct input of MPs to marine, however, is still unknown and it is difficult to trace the original sources of particles because of their small size (Jambeck et al., 2015). The composition of MPs in terms of particle morphology and polymer types may provide indications on their origins and former uses (Isobe et al., 2014; Takada and Tanaka, 2016).

Resin pellets, a polystyrene (PS) polymer about 2.5–5 mm in diameter, are the “industrial raw material” of consumer plastic products and classified as primary microplastics. They are the first type of MPs reported and enter the ocean through wastewater discharge from a plastic-producing/processing plant to a river/estuary (Colton, 1974). But with the strengthening of spilled pellets recapture and management in the plastic industry, the average concentration of resin pellets decreases significantly from the year of 1986–2008 (Law et al., 2010). As a result, MPs in the ocean are mostly from fragments and fibers in recent years, whereas pellets are only a small fraction (Browne et al., 2010; Eriksen et al., 2014; Reisser et al., 2013).

Photodegradation and oxidative and hydrolytic degradation can make plastics fragile and suffer from mechanical breakdown in the ocean (Feldman, 2002; Wagner et al., 2014). Morét-Ferguson et al. (2010) observed most fragments with characteristics of deterioration such as brittleness and rough edges/cracks from collected MP debris samples, proving that those MPs are secondary microplastics degraded from existed plastics in the ocean. Microfibers originate from the degradation of large plastic items of shipping activities, fishing equipment, recreation and offshore industries some of which may also arise from the washing of synthetic textiles rather than fragmentation in coastal areas (Browne et al., 2011; Dubaish and Liebezeit, 2013).

Even though the sources of MPs are known, the precise origin of an

individual MP cannot be targeted currently (Lusher et al., 2015; Woodall et al., 2014). Additionally, since MPs are mobile, both affected by the source of local and external pollution, they have been likely reached the Arctic in the long run at sea and transported over long distances (Lusher et al., 2015).

3. MPs in freshwater

3.1. Occurrences

About 80% of marine plastic debris come from terrestrial sources (Jambeck et al., 2015; Mani et al., 2015; Wagner et al., 2014), including inadequately disposed plastics introduced through freshwater transport. It is well recognized that not only MPs are present in freshwater but also their contamination is as severe as that in the ocean (Wagner et al., 2014). The widespread occurrence of MPs has been documented in freshwater ecosystems (Table 2). According to the beach cleanup report, plastic debris from garbage produced by human has been found along the Great Lakes' shorelines (Driedger et al., 2015). An expedition of the Laurentian Great Lakes of the United States indicates that all samples except one contain plastics and the frequency of occurrence among Lakes Superior, Lake Huron and Lake Erie are 100%, 87.5%, and 100%, respectively (Eriksen et al., 2013). Baldwin et al. (2016) collected samples from the 29 Great Lakes tributaries in the United States and found that MP concentrations are from 0.05 to 32 items/m^3. In Swiss, among top six large lakes, all 39 surface samples collected contain MPs and they are found in all investigated matrices (beach sediments, lake and river surfaces) (Faure et al., 2015). Even at a remote lake, Lake Hovsgol, MPs are observed in all nine pelagic survey transects (Free et al., 2014). Values of MPs in surface water of other lakes are 105,503 particles/km^2 (Lake Erie), 5390 particles/km^2 (Lake Superior), 2779 particles/km^2 (Lake Huron), and 20,000 particles/km^2 (Lake Hovsgol) (Eriksen et al., 2013; Free et al., 2014).

Several studies have reported the worldwide occurrences of MPs in major rivers (Mani et al., 2015), urban rivers (Moore et al., 2011), and estuarine rivers (Yonkos et al., 2014). In surface water samples collected from four estuarine tributaries of the Chesapeake Bay in the United States, MPs are found in 59 out of 60 samples (Yonkos et al., 2014). MPs are also found in all samples at 11 locations along the Rhine River with a length of 820 km (Mani et al., 2015). Moore et al. (2011) found that MP numbers are ranged from 0.01 to 12.9 particles/L in Los Angeles River, San Gabriel River, and tributary Coyote Creek. In a highly urbanized river of Chicago, concentrations of MPs in surface water reach 730,341–6,698,264 items/km^2, comparable or even higher than the values of oceans and the Great Lakes (McCormick et al., 2014). In Yangtze River of China, MP concentrations are 3,407,700–13, 617, 500/km^2 in the main stream and 192,500–11, 889, 7 00/km^2 in the estuarine areas of four tributaries (Zhang et al., 2015). As shown in Table 2, the concentrations of MPs in freshwater system are highly variable, likely owing to several factors including particle size, human population density, economic and urban development, waste management, and hydrological conditions (Eriksen et al., 2013; Free et al., 2014; Moore et al., 2011; Zbyszewski and Corcoran, 2011). It can also be concluded from Table 2 that more than 72% of MP particles are in the smallest size fraction (0.33–1 mm) in terms of their size distribution. The inverse relationship between MP concentration and particle size has been observed in many studies of rivers, lakes, and oceans (Table 2).

In general, the higher population density the site has, the more MPs will be found (Eerkes-Medrano et al., 2015). Eriksen et al. (2013) reported greater concentrations of plastics from surface trawls in Lake Erie near the population centers (i.e., Buffalo, NY; Erie, PA; and Cleveland, OH in the United States) compared to significantly less populous regions of the Lakes Huron and Superior. Mani et al. (2015) reported the maximum concentration of 3.9 million particles/km^2 in metro region of Rhine-Ruhr in Germany. In twenty-nine Great Lakes tributaries (Baldwin et al., 2016) and four estuarine tributaries within the Chesapeake Bay, the significantly positive correlation was also found by Yonkos et al. (2014). However, some distant areas with low population densities are also found with heavy MP pollution. In a remote mountain area of the Lake Hovsgol in Mongolia, MP concentration reaches 44,435 particles/km^2 due to improper waste management (Free et al., 2014). Hydrology may also affect the concentrations of MPs (Baldwin et al., 2016; Mani et al., 2015; Yonkos et al., 2014). In urban and non-urban watersheds, the concentrations of MPs are higher during runoff events than those under low-flow conditions, but the relationship has not been statistically verified.

3.2. Characteristics

Because PP and PE are widely used and less dense than water, they are still the two main MPs found in freshwater (Table 2). The types of MPs in freshwater are different from those in the ocean, due to the differences in pollution sources, location, and hydrodynamic conditions (Eerkes-Medrano et al., 2015). In the Taihu Lake of China, fiber is the most dominant MP component across all sample types with a proportion of 84% ($p < 0.01$) (Su et al., 2016). Fragments and fibers are accounting for 60% of all types of MPs in remote areas of the Lake Hovsgol in Mongolia (Free et al., 2014). In some rivers, spherical shape MPs such as spherules, pellets and microbeads are often the dominant species (Mani et al., 2015). On the surface water along the Rhine River, spherules make up almost 60% of the total MPs with a size range of 300–1000 μm (Mani et al., 2015). The freshwater pellets are shown similar characteristics to those of marine but apparently smaller than marine pellets (Carpenter et al., 1972). Microbeads are one of the primary sources of MPs in freshwater in granular or spherical shape ranging in size from roughly 8 μm to 2 mm (Napper et al., 2015; Takada and Tanaka, 2016) and more than 90% of microbeads in freshwater are made from PE (Takada and Tanaka, 2016). Eriksen et al. (2013) found microbeads of < 1 mm in surface water of the Laurentian Great Lakes in the United States. The fact that MPs in freshwater are different from those in the ocean suggests that they may have different sources. In addition, it has been reported that spherical shape MPs are hard to be transported to long distance and rivers are potentially their major sinks (Faure et al., 2015).

3.3. Sources

The origin of freshwater MPs can be classified into point sources, non-point sources, and tributaries. Freshwater is generally closer to the pollution sources and has smaller waterbody area than the ocean, its MPs thus are affected more by local sources (Mani et al., 2015). Mani et al. (2015) suggested that MP types and concentrations are diverse in freshwater because of their different sources and sinks. Baldwin et al. (2016) pointed out that there is a positive relationship between urban watershed attributes and MP concentrations. In the Rhine and Danube Rivers, most of the MPs are spherules, the PS spherules possibly originate from the plastic wastes of product manufactures discharged to the water treatment systems, while the PE spherules (microbeads) are from personal care products (PCPs) (Lechner and Ramler, 2015; Mani et al., 2015). These particles are commonly found near wastewater treatment plants (WWTPs) and plastic manufacture plants (Colton, 1974; Lechner et al., 2014). WWTPs outlets are found to be one of the most important point sources of MPs (Dubaish and Liebezeit, 2013; Eerkes-Medrano et al., 2015; McCormick et al., 2014). In the Chicago North Shore Channel, the concentration of MPs at the downstream sampling site is 9.2 times higher than that at the upstream site (McCormick et al., 2014), suggesting WWTP effluent is a point source of MPs. Fibers and pellets in freshwater are traced back to wastewater discharges associated with synthetic textiles and personal care products, respectively (McCormick et al., 2014). According to studies related to the fate of MPs during the wastewater treatment process, the grease removal

stage, solid skimming, sludge settling, and other processes can settle out MPs into sludge efficiently. In secondary plant, the removal efficiency can be up to 98.41% with 0.9–250 items m^{-3} of outflow concentration, while those numbers are changed to 99.9% and 0.002 items m^{-3} in tertiary plants with 0.598 mm of average size (Carr et al., 2016; Murphy et al., 2016). Carr et al. (2016) claimed that tertiary wastewater effluent is not a significant source of MPs to receiving water because of the efficient removal rate. However, the volume of effluent of WWTPs is large so that many small size MPs may not been captured by the treatment process (Chang, 2015; Fendall and Sewell, 2009). According to the statistics, 8 trillion microbeads are emitted into aquatic habitats per day in a final effluent of 0–7000 microbeads m^{-3} in the United States (Rochman et al., 2015). Even if the WWTPs are effective, sludge may not be the final sink for MPs. During the sludge treatment and disposal, MPs may be resuspended and become airborne, being spread in the terrestrial ecosystems (Rillig, 2012a).

Non-point source is another important pollution source of MPs, associated with runoffs, accidental effluent, and wastewater discharge and waste management in underdeveloped regions (Free et al., 2014; Morritt et al., 2014). After a rain, small MPs (1–4.75 mm) are 16 times more abundant in the Los Angeles River than large plastic particles (> 4.75 mm) (Moore et al., 2011). In the 29 Great Lakes tributaries, higher MP concentrations are found during runoff-event than under normal conditions (Baldwin et al., 2016). For some underdeveloped or remote regions, no WWTP is available and recycle and disposal of trash are not well managed, which may become non-point sources of MPs in freshwater. For example, the relative high concentrations of MPs in the Three Gorges Reservoirs (TGR) in China are mainly from non-point sources and carried to the reservoir by surface runoffs as most areas within the TGR are underdeveloped (Zhang et al., 2015). The Lake Hovsgol in Mongolia, a remote mountain lake with low-density populations, is another examples of non-point source pollution of MPs due to improper waste management (Free et al., 2014).

It can be concluded that MPs in freshwater including lakes and rivers are mainly from the discharges of WWTPs along the shore given that pellets/microbeads are in large portion. However, secondary MPs from non-point sources cannot be ignored in freshwater. It is still uncertain about the distribution and proportion of MP sources; hence, further investigations are necessary to obtain detailed information on sources of MPs in freshwater.

4. MPs in other media

4.1. MPs in sediments

4.1.1. Bottom sediments

When MPs lose their buoyancy, they can sink and accumulate in sediments. Since Thompson et al. (2004) reported the presence of MPs (1.6 μm–5 mm) in marine sediments in 18 locations across the UK, many reports have confirmed the severe MP sediment pollution worldwide, especially in coastal shallow water regions (Browne et al., 2011; Van Cauwenberghe et al., 2015). Even in the remote and largely unexplored deep-sea sediments, MPs have been identified with concentrations up to 2000 particles m^{-2} (Fischer et al., 2015; Schlining et al., 2013; Van Cauwenberghe et al., 2013). The subtidal or continental shelf areas near shores are also proved to be the hot pots of MP pollution. Among those areas, MP concentrations can reach 20–3320 items/L in Sweden (2007), 97.2 items/kg dry sediment in Belgium (Claessens et al., 2011), 672–2175 items/kg dry sediment in Italy (Vianello et al., 2013), and 10 items/kg sediment in Portugal (Frias et al., 2016).

MP pollution in freshwater sediments has also been documented, showing the same level heavy pollution in a wide range of distribution and high concentration. Reports from UK (Horton et al., 2017b), Italy (Imhof et al., 2013), Germany (Klein et al., 2015), Canada (Castañeda et al., 2014), and China (Di and Wang, 2018) have illustrated the MP occurrences in freshwater sediments. Imhof et al. (2013) examined the occurrence of MPs in beach sediments in the subalpine Lake Garda, Italy and reported concentrations of 108 (south shore) to 1108 (north shore) particles/m^2. Klein et al. (2015) measured the shore sediment of the Rhine Main area rivers in Germany and suggested that 100% of the sampled sediments have MP particles with mass fractions of up to 1 g/kg or 4000 particles/kg. Castañeda et al. (2014) studied MPs in the sediments of the St. Lawrence River and reported that the mean densities across sites are 13,832 items/m^2, with size of 0.40–2.16 mm in diameter. Horton et al. (2017a) investigated MPs in sediments of the River Thames basin in UK, and found that large MP particles (1–4 mm) at all four sites with the highest number of particles of 660 particles/kg. In a study of the Three Gorges Reservoir (TGR), China, the MP concentrations in the sediments are from 25 to 300 particles/kg (Di and Wang, 2018).

Because MPs in sediments are mainly from their settlements, the factors affecting sediment MP concentrations are as same as the corresponding surface water. Taking the population density as an example, in a study analyzing sediments from 18 locations representing 6 continents, Browne et al. (2011) demonstrated a positive relationship between MP concentration and human population density. Naji et al. (2017) found that the sediments with the highest number of MPs are from sites in the vicinity of highly populated centers and municipal wastewater discharges. In an investigation of the three Gorges Reservoir in China, however, Di and Wang (2018) found more MPs in sediments in the countryside than those in the urban areas and concluded that MP concentrations in sediments were not directly proportional to those in the corresponding surface water. The distribution of MPs in sediments, however, is still largely unknown due to the lack of uniformity among studies (Hidalgo-Ruz et al., 2012).

4.1.2. Beach sediments

It is important to choose the appropriate site or zone for MP pollution assessment in coastal regions. Browne et al. (2011) reported MP contaminations at 18 beaches (1 cm depth) in six continents from the poles to the equator and reported the concentrations of MPs in sediments ranging from 2 fibers/L in Australia to 160 fibers/L in the United Kingdom (UK) and Portugal. Van Cauwenberghe et al. (2015) reviewed over 100 studies on MPs in sediments and found that the concentrations of MPs ranging from 60 (in Brazil) to more than 285,673 (in South Korea) items/m^2 in sandy beaches and from 2.3 to more than 8000 (in Canada) items/kg dry sample in sediments. Investigations conducted by Van Cauwenberghe et at. (2015) show a broad MP distribution at coastal aquatic systems and a peak concentration at the harbor in sediments of coastal harbor, beach and sublittoral areas of Belgium. MP concentrations in these sites are significantly different due to the factors including freshwater inputs, urban discharges, human population density, urbanization, industrialization, aquaculture farming, and monitoring methods (Vianello et al., 2013). According to a study of vertical distribution, Martin et al. (2017) found that 97% of recovered MPs are in sediment shallower than 2.5 cm. The authors also pointed out that because of the force of shelf edge oceanography near the edge of the Rockall Trough, MPs might be accumulated, and a hotspot would be shaped in shelf break zones due to the canyon feature.

4.1.3. Characteristics and sources

MPs recovered from sediments show different shape, polymer types, colors, and physical forms, suggesting a wide range of sources (Martin et al., 2017). Microbeads/pellets, fragments and fibers are major MPs in sediments and they are mainly PP, PE, and PS. For the sediment in the freshwater and marine beaches, the MPs are mainly coming from coastal industrial and municipal wastewater discharges and direct plastic disposals. Thompson et al. (2004) conducted a study on MPs in beach sediments in Plymouth, UK and found that the MPs are granular and fibrous fragments belonging to the secondary MPs. However, Castañeda et al. (2014) detected high concentrations of microbeads in

the sediment of the Saint Lawrence river, Canada, which are primary MPs from municipal and industrial sewage effluents. Horton et al. (2017a) observed a high number of MPs in the River Thames basin (UK) at the downstream of a storm drain outfall receiving urban runoff and found that the MPs are mainly from thermoplastic road-surface marking paints. In deep sea, which is distant from human activities, most of the sediment MPs are microfibers classified as the secondary MPs from large plastic fragmentation (Taylor et al., 2016).

4.2. MPs in soils

Previous studies on MPs have focused on the aquatic environment heavily and only few of them have paid attention to soils. Occurrence of MPs in soils, however, has been documented in the literature. Zubris and Richards (2005) identified synthetic fibers in a soil applied with organic wastewater sludge for 15 years in the United States. Rillig (2012b) reviewed the potential of soil MP contamination and suggested that besides waterbody and sludge, soil is another sink for MPs due to solid waste landfill and dump and sludge fertilizer. Since then, many studies have been conducted to explore the effects of soil MPs on microbial and earthworm activities (Huerta Lwanga et al., 2016; Huerta Lwanga et al., 2018; Lwanga et al., 2017; Rillig, 2012b; Yang et al., 2018). The existence of MPs may also change soil physical properties, such as porosity and aggregate structure (Rillig, 2012b; Zhang et al., 2015). Liu et al. (2017) found that MP addition can stimulate enzymatic activity to activate organic C, N, and P pools and thus promote their accumulation in the dissolved phase. Anyhow, agricultural and urban soils can be more important environmental reservoirs of MPs even than the ocean (Hurley et al., 2018). Bläsing and Amelung (2018) also classified the sources of soil MPs in to the primary and secondary. Application of sewage sludge on farm soils is one of the largest sources of primary MPs (Nizzetto et al., 2016). Nizzetto et al. (2016) estimated that annually 63,000–430,000 and 44,000–300,000 tons of MPs may enter farmland soils through manure application in Europe and North America, respectively. Secondary MPs in soils can result from abrasion of plastic debris, including the plastic mulch and incidental plastic debris, at soil surfaces or inside the soil profile (Rillig, 2012b). Studies have already reported the formation of MP residues of various sizes originating from mulching (Briassoulis et al., 2015). Moreover, very small particles or fibers can be spread further by airborne transport and atmospheric deposition into soils (Bläsing and Amelung, 2018; Dris et al., 2015).

Because of the lack of efficient analytical methods to isolate MPs from complex organic soil matrix (Hurley et al., 2018), it is still hard to fully characterize and quantify MPs. Fuller and Gautam (2016) integrated pressurized fluid extraction (PFE) with Fourier-transform infrared spectroscopy (FTIR) to identify and quantify MPs in different kinds of soils in an Australian industrial area to show the existence of MPs in terrestrial media. Nevertheless, the characterization of MPs in soils has not been developed comprehensively yet because of technical limitations.

4.3. MPs in air

Some researchers have indicated that the atmospheric deposition is a potential source of MPs in the aquatic environment (Bläsing and Amelung, 2018; Free et al., 2014). MPs may be blown out from surfaces of poorly managed landfills or streets and stay in the air. Dris et al. (2016) investigated the atmospheric fallout of MPs in two different urban and sub-urban sites of Paris (France) and reported that the atmospheric fallout of MPs (mainly fibers) is between 2 and 355 particles/m^2/day. In a lately study, Dris et al. (2017) found MPs in both indoor and outdoor air and indoor settled dust. Indoor MP concentrations range between 1.0 and 60.0 fibers/m^3 and outdoor MP concentrations are significantly lower ranging between 0.3 and 1.5 fibers/m^3. The deposition rate of the MPs in indoor environments is between 1,586 and 11,130 fibers/day/m^2. About 33% of MP fibers in the indoor environment contain petrochemicals (predominantly polypropylene). These studies suggest that the atmospheric phase contains MPs that can lead to human exposure. The inhalation of MP particles and fibers have been reported as a health risk too (Dris et al., 2017).

5. Environmental fate and transport of MPs

5.1. Transport

To understand the environmental fate of MPs, it is essential to know their transport behaviors. It is well recognized that water, air, and soil are common pathways for MP transport (Fig. 3). In aquatic environment, area of water surface, depth, prevailing wind, surface current and density of particles are all important factors determining MP transport (Fischer et al., 2016; Free et al., 2014). The density of most MPs is lower than that of fresh or sea water (Andrady, 2011), so that MPs are often buoyant at the water surface, transported with water along rivers and into oceans, which is known as surface transport (advective transport). Besseling et al. (2017) calculated the transport of nanometer to millimeter sized spherical particles (i.e. microbeads, 100 nm to 10 mm) in freshwater systems and found the 99% retention distance (RD99) to be around 200 km and up to > 900 km for nanoplastics and MPs, respectively. They suggested that the intermediate size class of MP may be preferentially transported downstream.

In general, the bigger plastics are, the easier they will drift in the uppermost layer. Hence, MPs are less affected by stoke drift resulted from surficial water's wind waves so that they are more likely to be carried offshore (Isobe et al., 2014). Additionally, turbulence in the upper-water layer can vertically mix buoyant MPs, therefore, there are some vertical transports in the water column and a better understanding of this type of transport of buoyant plastics is important. Reisser et al. (2015) found that MPs with lower rise velocities are more susceptible to vertical transport. Because investigations on MP vertical transport are heavily relied on observations, current multi-level plastic sample technologies are in low-resolution and haven't been well-developed (Isobe et al., 2014; Kukulka et al., 2012).

MPs with higher density are more likely to retain in soils and be transported to deeper soil layer finally; whereas those MPs with lower density are more susceptible to wind and surface runoffs and to reach surface aquatic and terrestrial systems furthermore (Zylstra, 2013). Earthworms can be a significant transport agent of MPs in soils through many pathways such as casts, burrows, egestion and adherence to the earthworm exterior, resulting vertical transfer of MPs in soil profiles (Lwanga et al., 2017; Rillig et al., 2017).

On the other hand, MPs may enhance the transport of persistent, bio-accumulative, and toxic substances. MPs can be a transport vector of toxic metals (Brennecke et al., 2016; Rochman et al., 2014) and persistent organic pollutants (POPs) (Gouin et al., 2011; Rochman et al., 2013). Some of these compounds are added into MPs during manufacture, while others adsorb on MP surfaces. Previous studies have demonstrated the transfer of contaminants from MPs to organisms (Avio et al., 2015; Browne et al., 2013; Chua et al., 2014; Teuten et al., 2009).

5.2. Fate

It is generally believed that once MPs reach waterbody, they will enter the ocean ultimately. It has been estimated that about 70% of marine trash are settled down to the sediment at the bottom of ocean. The half of the remaining 30% floats on the surface seawater (15%) and the other half is among coastal areas (15%) (UNEP, 2005). Because most MPs are lighter than sea water and are buoyant at the sea surface, the most extensive spatial pattern in sea surface MPs is their accumulation in large-scale subtropical oceanic gyres, where convergent ocean surface currents concentrate and retain debris over long time periods

(Eriksen et al., 2014; Goldstein et al., 2012; Wilber, 1987). Cózar et al. (2014) estimated the range of total plastic load from 7,000 to 35,000 tons in the ocean surface layer worldwide. The major losses of small plastics in the seawater surface have been determined by measuring the size distribution of MPs at different places all over the world (Cózar et al., 2014; Eriksen et al., 2014). Kukulka et al. (2012) claimed that there is an significant underestimation of total plastic concentration in the ocean by traditional measurements. Reisser et al. (2015) studied the vertical distribution of MPs in water column and explained a fraction of this "missing" plastic under the sampled surface layer (0–0.5 m). They pointed out that vertical mixing can affect the size distribution of plastics floating at the surface because smaller plastics are more susceptible to vertical transport. Besides the accumulate in oceanic gyres and shallow water sediments, deep-sea sediments are a potential sink of MPs, explaining the missing fraction. Woodall et al. (2014) found that fibers are up to four times of magnitude more plentiful (per unit volume) in sediments than surface seawater in the Atlantic Ocean, the Mediterranean Sea, and the Indian Ocean. Obbard et al. (2014) suggested that the polar sea ice may also represent a major historic global sink of MPs.

Weathering processes including photooxidation, oxidative, hydrolytic degradation, and biodegradation processes can strongly affect the fate of plastic debris in the aquatic environment. Those processes turn plastic fragmentation into smaller particles (Barnes et al., 2009; Lambert and Wagner, 2016) and simultaneously change the condition of the MPs and their hydrodynamic behaviors (Ter Halle et al., 2016). Hetero-aggregation and biofilm formation also play important roles in affecting the fate of aqueous MPs (Rummel et al., 2017; Woodall et al., 2014). The hetero-aggregation and biofilm formation may cause an increase of MP density and a decrease of their buoyancy (Lagarde et al., 2016); and smaller MPs tend to reach a significant precipitation density in a faster way (Chubarenko et al., 2016). A higher density than the ambient water indicates sedimentation given that the sinking rate reflects particle size and density (Long et al., 2015). Meanwhile, biofilm formation can make MP become sticky because of the extracellular polymeric substances matrix, promoting the formation of hetero-aggregates. Additionally, the downhill transport of MPs may be improved due to the discharge of fecal pellets of zooplankton (Cole et al., 2016; Gorokhova, 2015).

It is important to consider temporary and permanent sinks of MPs. Possible sinks for MPs include fragmentation, sedimentation, shore deposition, and ingestion by organisms (Law et al., 2010). These sinks, however, are dynamic. For example, MPs can be trapped in sediments over a long time. Affected by wave action, currents or bioturbation, and some other disturbances, those trapped MPs may be more readily resuspended from bottom sediments than larger plastic debris because of their smaller size and lower density compared to natural sediments (Kershaw and Rochman, 2015). Ingested by organisms, MPs may either be excreted as waste or retained/translocated into tissues, making trophic transfer and accumulating in food chain or more advanced organisms through food chain. The ingestion rate and the fate of ingested MPs are still largely unknown.

5.3. Fate and transport models

The study of MPs in environment is still in an early stage. Models that simulate MP fate and transport are crucial to understand their environmental impacts. Model-predicted environmental concentrations of MPs are important because there are insufficient data on actual concentrations (Kapustka, 2008). Although many places show the existence of large number of MPs worldwide, the detection methods of MP particles in natural samples are still in their infancy. The lack of identical standard, the variance of methods, and the negative comparability of data make the situation worse. For example, present methods are still insufficient to detect smaller size of MPs such as nanoplastics (Gigault et al., 2016; Koelmans, 2015; Lambert and Wagner, 2016; Song et al., 2015). Researchers thus have considered applying mathematical models to improve current understanding of MPs in the environment.

Current MP modeling approaches mostly rely on applying exiting models to predict the distribution and the fate and transport of surficial MPs. Models of engineered nanoparticles (ENPs), such as NanoDUFLOW, have been adapted for modeling the fate and transport of MPs in freshwater (Besseling et al., 2017; Quik et al., 2015). Law et al. (2010) used a numerical model based on a drifter statistical prediction to determine the distribution pattern of MPs. Lebreton et al. (2012) suggested that plastic pathways in ocean can be represented by the Lagrangian particle trajectories. MP prediction models have been developed based on the theory and revealed five main sites of drifter aggregation, located in the subtropics and maintained by conveying the Ekman currents (Lebreton et al., 2012; Maximenko et al., 2012). Additionally, because former studies of MPs mainly focus on the North Atlantic and the North Pacific accumulation zones, there is a lack of data of other areas. To quickly understand global abundance and weight of floating plastics, models have been used to estimate global accumulation number (Eriksen et al., 2014; Van Sebille et al., 2015). Van Sebille et al. (2015) summarized existed detection datasets and three different ocean circulation models, applying spatially interpolation to estimate the accumulated number of MP particles in 2014, ranging from 15 to 51 trillion particles and weighing between 93 and 236 thousand metric tons. Eriksen et al. (2014) predicted a dramatic decrease of MPs in seawater surface using an oceanographic model of buoyant plastic debris distribution. However, because of the scarcity of data, the difference of investigation methods and model formulations, and fundamental knowledge gaps in sources, transformations and fates of microplastics in the ocean, there are order-of-magnitude discrepancies in calculated results generated from models. For example, Lebreton et al. (2018) calibrated a model with data from multi-vessel and aircraft surveys, estimated that the "Great Pacific Garbage Patch" (~79 k tons) is nearly 16 times higher than that of one study used net trawl data only (Cózar et al., 2014) and 4 times higher than that of another assessment (~21 k tons) that combined net trawl data with vessel-based visual surveys (Eriksen et al., 2014).

Good models of MP fate and transport should consider all relevant processes in their governing equations; however, simplifications are also necessary to avoid over parameterization. The trade-off between model complexity and simplification is very important to the development of models of MP fate and transport, especially with respect to large-scale models that involve uncertainty and data scarcity. To overcome the uncertainty and data absence in large scales of global prediction, Critchell and Lambrechts (2016) selected coastal zones, developed a plastic oceanographic model (an advection-diffusion model) to study the fate and transport of plastics in estuarine and coastal waters. The model was designed to quantify the relative effects of important physical processes (e.g., settling, fragmentation, re-suspension/re-floating, and topographic effects on the wind) on plastic accumulation based on the Second-generation Louvain-la-Neuve Ice-ocean Model (Lambrechts et al., 2008). The authors found that the physical characteristic of the source location has the largest effect on the fate and transport of MPs. In addition, the diffusivity, fragmentation rate, and relationship between debris re-suspension from beaches and the wind shadow created by high islands also have dramatic impacts on the modeling results; whereas settling, wind drift velocity and other processes play less important roles.

6. Environmental impacts of MPs

6.1. Toxic effects

MPs present an increasing threat to the environment and the ecosystems because of their potential toxicity as well as their durability and persistence (Hidalgo-Ruz et al., 2012; Lusher et al., 2014). In addition, several studies have suggested the potential roles of MPs as vectors of

other toxic chemical contaminants (Avio et al., 2015; Browne et al., 2013; Chua et al., 2014). To understand the toxicity of MPs thus is crucial to the assessment of their environmental impacts.

MPs are easier to be ingested by a wide range of organisms due to the small size compared to macro-plastics (Barnes et al., 2009; Law and Thompson, 2014). MP uptakes by organisms such as lugworms, mussels, amphipods, barnacles, sea cucumbers, and fish have been documented, but their toxicity has not yet been fully understood yet (Browne et al., 2008b; Foekema et al., 2012; Thompson et al., 2004). During the process of ingestion, the physical components and their toxic chemicals of such small MPs can make harmful effects to the organisms. MPs may present a physical hazard in a similar way to a large item by internal abrasion and cogging feeding appendages or the digestive system (Cole et al., 2013; Derraik, 2002; Laist, 1997; Wright et al., 2013). Experimental exposures (e.g., blue mussels (*Mytilus edulis*)) have shown that MPs can also be taken up from the gut into other body tissues (e.g. haemolymph) and cause increased granulocytomas and decreased lysosomal membrane stability (Browne et al., 2008a; Von Moos et al., 2012). MP ingestion can also reduce algal ingestion rate in copepods (*Centropages typicus*) and decrease the feeding activity and weight of lugworms (Besseling et al., 2012; Cole et al., 2013). Results from invertebrates are consistent with findings of reduced feeding and physical condition in seabirds (Ryan, 1988; Spear et al., 1995). More importantly, after ingestion, MPs may transfer and release toxic chemicals (Rochman et al., 2013; Wardrop et al., 2016), hence, their toxic effects may be more severe with chemical injury compared to physical damage (Lithner et al., 2011; Rochman et al., 2013). These chemical injuries have been related with several adverse effects including carcinogenic and endocrine disrupting effects (Teuten et al., 2009), decreased fish populations (McKinley and Johnston, 2010), reduced evenness and richness of species (Johnston and Roberts, 2009), decreased growth and reproduction on the freshwater amphipod (Au et al., 2015), and significant effect on fitness of lugworm (Besseling et al., 2012).

Due to their relatively large specific surface area, MPs can sorb and concentrate many organic and inorganic chemical contaminants, introducing indirect toxicity (Eerkes-Medrano et al., 2015; Rochman et al., 2013; Teuten et al., 2009). MPs keep the original characteristics of plastics, containing a multitude of chemical additives such as phthalate plasticizers, brominated flame retardants, antioxidants, processing chemicals, colorants, and pigments (Eerkes-Medrano et al., 2015; Teuten et al., 2009). Some hydrophobic organic contaminants such as POPs have a greater affinity for the hydrophobic surface of MPs, and they may be concentrated on MPs at up to 6 orders of magnitude higher than those in ambient seawater (Browne et al., 2007; Hirai et al., 2011; Mato et al., 2001). The concentration of POPs in pellets of ocean is between 1 and 10,000 ng/g globally (Hirai et al., 2011; Ogata et al., 2009). For polychlorinated biphenyls (PCBs), their global concentration is from 4 to 980 ng/g of plastic pellets (Mato et al., 2001; Rios et al., 2007). Different kinds of polymers of MPs may introduce different indirect toxicity. Several POPs, PCBs, organo-halogenated pesticides, nonylphenol, PAHs, and dioxins have been detected in beach plastic pellets at various locations (Endo et al., 2005; Filella and Turner, 2018; Heskett et al., 2012; Hirai et al., 2011; Ogata et al., 2009). However, it has been proposed that many inorganic additives, such as heavy metals, may be more toxic compared to those organic pollutants (Filella and Turner, 2018).

Only few studies have investigated the ecological effects of MPs. To date, the ecological impacts of MP ingestion is still poorly understood (Browne, 2015; Moore, 2008; Wright et al., 2013). There is an increasing concern that the accumulation of MPs may affect the functioning of marine ecosystems; however, the influence can only be inferred by research results without direct field evidence. Because of their limited ability to regulate their internal environment, eggs, embryos, and larvae of aquatic organisms are particularly vulnerable to MPs (Sussarellu et al., 2016). In early stage of growth, because of the existence of strong selection driven by antipredator (Bailey and Houde, 1989; Leggett and Deblois, 1994) and proximate factors (e.g. feeding history), there may be a reduced food intake and population growth when they cannot actively avoid MP intake (Lönnstedt, 2012). Besides, MPs can kill or injure ecologically (e.g., primary tropic levels or keystone creatures) and commercially important species, including mussels, zooplankton, salt-marsh grasses, and corals (Browne et al., 2008a; Cole et al., 2013; Uhrin and Schellinger, 2011). Due to the size similarity of MPs to sediments and some planktonic organisms, they can be ingested by low trophic suspension, filter and deposit feeders, detritivores, and planktivores (Browne et al., 2008b; Graham and Thompson, 2009; Murray and Cowie, 2011). Because of the polymer density and size, MPs may be more available for pelagic filter feeders than sand or silt, which settle relatively fast. The high capacity of zooplankton species (Agasild and Nõges, 2005; Setälä et al., 2014) and mussels (Setälä et al., 2016) to ingest MPs suggests that filter feeders are most vulnerable to the exposure of suspended MPs (Scherer et al., 2017). At the bottom of the North Sea, lugworm is a powerful deposit feeder in the food chain (Thompson et al., 2004). Both feeding experiments and field surveys demonstrate the uptake of MPs by the lugworm (Besseling et al., 2012). MPs can also affect the growth of young creatures, the transfer of toxic substances among organism, and ecological balance. They can also influence local ecosystem through the transport of colonial microbes on MPs surface to a new environment.

Human populations can be exposed to MPs directly from the environment or indirectly through the food (Takada and Tanaka, 2016). Rochman et al. (2015) assessed the presence of MPs in fishes for human consumption and found MPs in 25–33% of individual fish and 55–67% of all species. Japanese anchovy is one of the most caught fish species and as a common food in Japan, it is typically eaten without removing the digestive tract. The field survey of Takada and Tanaka (2016) found around 2 pieces of MPs per fish on average in the anchovies and further confirmed that humans have been exposed to MPs. Furthermore, the annual dietary exposure for shellfish consumers in Europe can reach 11,000 MPs statistically (Van Cauwenberghe and Janssen, 2014).

Harm of MPs has been found in human. There is a disruption of cellular processes and tissues when patients' broken knee or hip joints have been substituted with plastic implants (Nuss and Rechenberg, 2008). Inhaled MP fibers taken up by the lung tissues can become associated with tumors (Pauly et al., 1998) and dispersive dyes originated from PE and acrylic fibers have been shown to cause dermatitis (Pratt and Taraska, 2000). Biological toxicity analysis shows that the potential toxicity from MP exposure can induce disturbance of energy and lipid metabolism as well as oxidative stress (Deng et al., 2017).

6.2. Bioaccumulation and bioavailability

Lower trophic creatures (e.g., zooplanktons and invertebrates) can ingest and accumulate MPs, realizing the trophic transfer and accumulation in food web. Nevertheless, only few studies focus on the bioaccumulation of MPs and their associate pollutants in organisms. It is most likely that there are interactions between organism and MPs because of the widespread presence of MPs in the environment (Collignon et al., 2012). MPs have been found inside the stomach (Romeo et al., 2015; Taylor et al., 2016), oral (Hall et al., 2015; Taylor et al., 2016) and ventilation areas (Watts et al., 2014) of organisms. They have also been identified in the gastrointestinal tracts of 36.5% in 10 species of fish from the English Channel in field surveys (Lusher et al., 2013). Besides the bioaccumulation of MPs in the digestive tract, feeding experiments have demonstrated that MPs can be translocated to accumulate in specific tissues and cells, such as in gills and guts of shore crab (*Carcinus maenas*) (Watts et al., 2014), in liver and gut of zebrafish, in stomachs of seabird, giant fish, and whale (Lusher et al., 2015; Romeo et al., 2015), in lysosomal system of mussel (Von Moos et al., 2012), and in the haemolymphand inside the haemocytes. In addition to being drawn into the gills of the blue mussel (*Mytilus edulis*), MPs are

found in stomach and then transported into digestive gland, showing an accumulation in lysosomal system after a 3-h exposure (Von Moos et al., 2012). Deng et al. (2017) reported the distribution and accumulation of MPs (5 μm and 20 μm) across mice tissues. Setälä et al. (2014) showed the transfer of plastic particles from mesozooplankton to macrozooplankton (10 mm, polystyrene). Farrell and Nelson (2013) illustrated the "natural" trophic level transfer of microplastic mussels (*Mytilus edulis*) to crabs (*Carcinus maenas*) and its translocation to haemolymph and tissues of a crab. When it comes to chemicals such as POPs adsorption on MPs, experimental exposure tests have demonstrated their transport and accumulation in fish tissues with MP ingestion (Rochman et al., 2013; Wardrop et al., 2016). It has been speculated that marine MPs may increase bioaccumulation of POPs in the food web. Besseling et al. (2012) found that the bioaccumulation of POPs by lugworm (*Arenicola marina (*L.)) may be significantly affected by PS. However, at present, there is still limited information of the impacts of MPs and the associated pollutants on food webs.

The overall impacts of MPs to organisms are also affected by their bioavailability (Cole and Galloway, 2015; Wright et al., 2013), which strongly relies on MP physiochemical properties such as particle size and polymer density (Scherer et al., 2017; Wright et al., 2013). The small size enhances MPs' bioavailable to be direct ingested by a wide range of organisms (Law and Thompson, 2014). Because their size fraction is similar to those of sediments and planktonic organisms, a planktivore can passively ingest MPs during normal feeding behavior or mistake them as a natural prey (Wright et al., 2013). In feeding experiments, Scherer et al. (2017) found that despite the intraspecific variability in feeding rates, uptake of 90 μm MP particles by *C. riparius* is significantly lower than that of 10 μm ones ($p < 0.01$). It has been concluded that the potential accumulation and bioavailability of MPs in the food chain increase when their sizes decrease. Watts et al. (2014) reported that the shore crab (*Carcinus maenas*) can ingest MPs by gill inspiration and mussel (pre-exposed) ingestion. The density of plastic particles may also affect their bioavailability in water column. For example, planktivores, filter feeders, and suspension feeders inhabiting the upper water column are likely to encounter positively buoyant with low-density plastics, such as PE, on the sea surface. But besides the impact of self-density of polymers, MPs are also influenced by biofouling and aggregation, either sinking below the sea surface, leading to the decrease of buoyancy (Lagarde et al., 2016; Rummel et al., 2017) or returning to the sea-air interface after defouling (Andrady, 2011). Furthermore, there is a difference of bioavailability of irregular shaped plastic particles or fibers (Kowalski et al., 2016; Ogonowski et al., 2016). Nevertheless, what makes scientists more interested is that there is a potential for MPs to enhance the bioavailability of pollutants adsorbed on MPs surface. The bioavailability of PCBs and pyrene in vertebrates/invertebrates can be changed by the presence of MPs (Avio et al., 2015; Oliveira et al., 2013). Sleight et al. (2017) applied gene expression in larval zebrafish to assess the bioavailability of phenanthrene (Phe) and 17α-ethinylestradiol after MP settlement and found Phe is 48% more bioavailable than the prediction of a linear sorption model. Unfortunately, it is difficult to assess the MP-enhanced bioavailability of pollutants given the presence of too many interacting factors including pH, salinity, chemical interactions, temperature and organism variables (e.g., respiration and digestive systematic function). For example, MP uptake of seabirds has been reported to elevate the amounts of PCBs and other POPs (Tanaka et al., 2013), but whether the increased quantity of these contaminants is associated with the ingested MPs is still unknown.

7. Conclusions and perspectives

As an emerging contaminant, MPs have been found almost everywhere in the environment indicating widespread distribution of their contamination. Based on the literature review, it can be concluded that MPs are ubiquitous in the environment, particularly in the marine and freshwater systems, and have already imposed negative impacts on the ecosystem. With the increase of global production and consumption of plastics, MP contaminations in the environment are expected to continue to rise and may cause severe damages to the environment and ecosystems. MPs have attracted increasing research attention recently, leading to much better understanding of their occurrences, transport, transformation, fate, and impacts in the environment. As indicated in this review, however, current research on MPs in the environmental is still in its infancy. Additional research investigations thus are needed to further study MPs in the environment at both the laboratory and field scales, especially in following areas: 1) degradation of marine plastic debris and formation processes and mechanisms of secondary MPs in the ocean; 2) stability, retention, transport, and transformation of primary and secondary MPs in WWTPs and surface and groundwater systems; 3) occurrences, fate and transport, and impacts of MPs in soils; 4) mathematical models of fate and transport of MPs in the environment; 5) environmental impact assessment of MPs in the environment; and 6) effects of MPs on the transport, fate, and toxicity of other chemical contaminants in the environment.

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