



Review

The chemical behaviors of microplastics in marine environment: A review

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ABSTRACT

Microplastics are widely existed in marine and coastal environments, which aroused global concern in recent years. This review mainly summarized the interactions of organic pollutants and metals with microplastics based on environmental monitoring results and laboratory results reported by literatures. Firstly, the type, properties, and distribution of microplastics in the environment were briefly reviewed. Secondly, the property changes of microplastics after degradation were discussed. Thirdly, the concentrations of pollutants on microplastics in global environments were summarized. Then the effect of the factors (e.g. types and properties of microplastics, types of pollutants, and environmental conditions) on the sorption behaviors of microplastics were discussed in detail. Finally, the influences of microplastics on marine organisms were briefly evaluated.

1. Introduction

Microplastics, which are generally defined as plastics debris less than 5 mm in size, have gained increasing interest in recent years because of their large amount (about 8 million tons discharged in the ocean each year) (Jambeck et al., 2015; Phuong et al., 2016; Li et al., 2016; Auta et al., 2017; Carbery et al., 2018). They are widely detected in marine and coastal environments, including sediments (Dekiff et al., 2014; Vaughan et al., 2017), beaches (Van et al., 2012), seawater (Woodall et al., 2014; Zhou et al., 2018), river water (Wang et al., 2017), and even polar regions (Kanhai et al., 2018). Microplastics pollution in environments has two sources: primary source and secondary source. Primary microplastics are the plastics pellets, beads, nurdles, fibers, and powders used as industrial materials, personal care and cleaning products additives (Gregory, 1996; Fendall and Sewell,

2009). Secondary microplastics derive from the degradation of microplastics under weathering/aging processes (Browne et al., 2007). Secondary microplastics are abundant in globe marine and coastal environments. Notably, further degradation of primary and secondary microplastics will change the properties of microplastics (such as color, surface morphological, size, crystallinity, and densities), which may influence their physical and chemical actions in environments (Rincon-Rubio et al., 2001; Lambert and Wagner, 2016; Zhou et al., 2018).

Microplastics are inclined to sorb and accumulate pollutants from surrounding water (Holmes et al., 2012; Hodson et al., 2017; Holmes et al., 2014; Brennecke et al., 2016; Karapanagioti et al., 2011; Fries and Zarfl, 2012; Bakir et al., 2014a; Li et al., 2018) because of their small size and large surface to volume ratio. The concentrations of pollutants on microplastics have spatial variability (Ashton et al., 2010; Holmes et al., 2012; Turner, 2016; Vedolin et al., 2017; Filella and

Abbreviations: Acronym, Meaning; ABS, Acrylonitrile butadiene styrene; PS, Polystyrene; PE, Polyethylene; PP, Polypropylenes; PET, Polyethylene terephthalate; PA, Polypropylenes; PVC, Polyvinyl chloride; SEM, Scanning electron microscopy; FE-SEM, Field emission scanning electron microscopy; PFO, Pseudo-first-order; PSO, Pseudo-second-order; ACE, Acenaphthene; ACY, Acenaphthylene; AMX, Amoxicillin; ANT, Anthracene; BaA, Benz [a] anthracene; BEZ, Benzene; BaP, Benzo [a] pyrene; BbF, Benzo [b] fluoranthene; BkF, Benzo [k] fluoranthene; CBZ, Chlorobenzene; CHR, Chrysene; CHR-d12, Chrys-ene-d12; CIP, Ciprofloxacin; cHex, Cyclohexane; EBM, Ethyl benzene; EBT, Ethyl benzoate; ETBE, Ethyl tert-butyl ether; FLT, Fluoranthene; FLN, Fluorene; FLO, Fluorene; HBCDs, Hexabromocyclododecanes; HCHs, Hexachlorocyclohexanes; HOCs, hydrophobic organic chemicals; MTBE, Methyl tert-butyl ether; MK, Musk ketone; MX, Musk xylene; NAP, Naphthalene; nHex, n-Hexane; OX, O-xylene; PFASs, Perfluoroalkyl substances; PFBS, Perfluorobutanesulfonate; PFBA, Perfluorobutanoic; PFDS, Perfluorodecanesulfonate; PFDA, Perfluorodecanoic; PFDaA, Perfluorododecanoic; PFHpA, Perfluoroheptanoic; PFHxDA, Perfluorohexadecanoic; PFHxA, Perfluorohexanoic; PFHxS, Perfluorohexasulfonate; PFNA, Perfluorononanoic; PFODA, Perfluorooctadecanoic; FOSA, Perfluorooctanesulfonamide; PFOS, Perfluorooctanesulfonate; PFOA, Perfluorooctanoic; PFPeA, Perfluoropentanoic; PFTeA, Perfluorotetradecanoic; PFTra, Perfluorotridecanoic; PFUnA, Perfluoroundecanoic; PHE, Phenanthrene; PCBs, Polychlorinated biphenyls; PAHs, Polycyclic aromatic hydrocarbon; PX, P-xylene; PYR, Pyrene; SDZ, Sulfadiazine; SMX, Sulfamethoxazole; SM, Synthetic musks; TAEE, Tert-amyl ethyl ether; TAME, Tert-amyl methyl ether; TC, Tetracycline; TOL, Toluene; AHTN, Tonalide; TMP, Trimethoprim; TRI, Triphenylene; TBC, Tris-(2,3-dibromopropyl) isocyanurate; TYL, Tylosin

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Turner, 2018). The concentrations of PCBs, PAHs and HCHs on microplastics depend on the proximity to pollution sources, i.e., sampling near the pollution sources are higher than those far from the pollution sources (F. Wang et al., 2018). The sorption behavior of microplastics is related to the properties of microplastics and pollutants (Holmes et al., 2012; Guo et al., 2012; Turner and Holmes, 2015; Zhang et al., 2018; Hu et al., 2017). PE has been reported to sorb more organic pollutants than other kinds of microplastics (Bakir et al., 2012; Wang and Wang, 2018a, 2018b). And aged microplastics are inclined to sorb more pollutants than the virgin ones (Brennecke et al., 2016; Müller et al., 2018). The polarity of both microplastics and contaminants plays a role in sorption. Li et al. (2018) demonstrated that the polar polymer PA with amide group had higher sorption capacities for polar antibiotic. Wang et al. (2019a, 2019b) also found that polar PS showed higher affinity for polar nitrobenzene. The nonpolar FOSA have stronger partition ability towards nonpolar PE (Wang et al., 2015). In addition, the environmental conditions (e.g. salinity and pH) would also influence the sorption capacities of pollutants on microplastics. Wang et al. (2015) reported that increase of salinity would increase the affinities of PE for PFOS. Holmes et al. (2014) found that the sorption of Cd, Co, Ni, and Pb on PE pellets increased with increase of pH values.

Microplastics could be ingested by many kinds of marine organisms (Cole et al., 2013; Desforges et al., 2015; Leslie et al., 2017; Hipfner et al., 2018). Therefore, the contaminated microplastics may not only cause mechanical damages to organisms (Wright et al., 2013; Avio et al., 2015), but also introduce contamination to marine organisms and to the marine food web (Bakir et al., 2012; Santana et al., 2017). As an increasing amount of laboratory studies and environmental monitoring studies have investigated the sorption behaviors of microplastics, and the effect of microplastics on marine organisms, it is useful to review the interactions between microplastics and pollutants.

The objectives of this review were (1) to briefly review the types, properties, and distribution of microplastics in marine and coastal environments; (2) to describe the changes in the properties of microplastics after degradation and the possible mechanisms involved; (3) to summarize the concentrations of pollutants in globe marine and coastal environments based on the environmental monitoring results from literature; (4) to discuss the influencing factors in the sorption of pollutants on microplastics based on the laboratory studies; and (5) to briefly discuss the potential effect of microplastics to marine organisms.

2. Types, properties, and distribution of microplastics

The compositions of microplastics in marine environments are mainly including PE, PP, PS, PVC, PA, and PET (Andrady, 2017; Hidalgo-Ruz et al., 2012). The properties of these microplastics polymers are depicted in Table 1 (Crawford and Quinn, 2017). The degree of crystallinity refers to the proportion of crystalline regions in polymers, where the polymer chains are aligned with each other. The degree of crystallinity directly influences the mechanical properties of polymers. Semi-crystallinity polymers are characterized with high strength and high fatigue resistance. In general, amorphous polymers

are soft and flexible, and exhibit poor strength and poor fatigue resistance. In addition, the amorphous regions of polymers include glassy domains and rubbery domains. Polymers change their physical form from glassy to rubbery state when the temperature is above its glass transition temperature (T_g).

Density of microplastics determines the distribution of microplastics in the water column. In general, PE and PP are floating microplastics, as their densities are smaller than water. PVC, PS, PET, and PA are denser than water, so they are inclined to sink in the water column. However, the properties, such as crystallinity and density of microplastics are not inherent characteristic properties, which are easy to change through weathering/aging process. These changes in microplastics properties are overviewed in the following Section.

Table 2 shows the distribution and abundance of microplastics in global environments based on current studies. As depicted in Table 2, microplastics are widely detected in inland lakes, estuaries, oceans, and even remote areas, such as Arctic Central Basin (Kanhai et al., 2018). Evidence suggests that microplastics in surface water are more abundant than that in the water column (Xiong et al., 2018). The small microplastics (0.02–1 mm) are more abundant than the large ones (1–5 mm) (Eo et al., 2018). Moreover, the abundance of microplastics in seawater and in sediments shows regional differences. For example, the abundance of microplastics in sediments is 37.1 ± 42.7 items/kg in Yellow Sea, China (Zhu et al., 2018), 250–300 items/kg in Edgbaston Pool, UK (Vaughan et al., 2017), 1739 ± 2153 items/kg in Xiangshan Bay, China (Chen et al., 2018), and 9×10^5 items/km² in Balearic Islands, Spain (Ruiz-Orejón et al., 2018). The concentrations of microplastics are lower in remote areas. For example, in Northwestern Pacific, the abundance of microplastics is 1.0×10^4 items/km² (Pan et al., 2018), in Atlantic Ocean, this value is 1.15 ± 1.45 items/m³ (Kanhai et al., 2017). Moreover, it is clearly that various units, such as item/m², n/m², items/km², items/m³, and items/kg are used to describe the abundance of microplastics (Table 2), which make it difficult to statistically compare the abundance of microplastics in global environments. Therefore, efforts should be paid to develop standard methods for statistics studies.

3. Properties changes of microplastics after degradation

Macroscopic plastics and microplastics in marine and coastal environments undergo various weathering/aging processes, such as solar exposure, thermal aging, bio-film growth, and oxidation (Andrady, 2017), which would result in the degradation of plastics polymers. Degradation refers to a series of chemical reactions that breaks the structures of plastics polymers, which is often classified into photo degradation, thermal degradation, biodegradation and thermos-oxidative degradation, according to the different weathering processes. Degradation leads to the fragmentation of macroscopic plastics waste and the introduction of secondary microplastics into environments. For primary and secondary microplastics, degradation mainly changes their physical and chemical properties, such as color, surface morphological, crystallinity, particles size, and density (Rincon-Rubio et al., 2001;

Table 1
Types and properties of microplastics (Crawford and Quinn, 2017).

Microplastics	Chemical formula	Degree of crystallinity	Glass transition temperature (T_g) (°C)	Density (g/cm ³)
PE	(C ₂ H ₄) _n	Semi-crystalline	–110	0.92–0.97
PP	(C ₃ H ₆) _n	Semi-crystalline (isotactic, syndiotactic) Amorphous (atactic)	(–49)–(–20)	0.88–1.23
PVC	(C ₂ H ₃ Cl) _n	Amorphous	60–100	1.15–1.70
PS	(C ₈ H ₈) _n	Amorphous	90	1.04–1.50
PET	(C ₁₀ H ₈ O ₄) _n	Semi-crystalline	73–78	1.30–1.50
PA6	(C ₆ H ₁₁ NO) _n	Semi-crystalline	–60	1.12–1.14
PA66	(C ₁₂ H ₂₂ N ₂ O ₂) _n			1.13–1.38

Table 2
Abundance and distribution of microplastics in marine and coastal environments.

Location	Abundance	Reference
South Carolina Estuaries	6.6–413.8 item/m ²	Gray et al., 2018
Sea of Marmara, Turkey	1.263 item/m ²	Tunçer et al., 2018
South Korea	0–2088 n/m ² (large microplastics: 1–5 mm) and 1400–62,800 n/m ² (small microplastics: 0.02–1 mm)	Eo et al., 2018
Qinghai Lake, China	0.05 × 10 ⁵ –7.58 × 10 ⁵ items/km ² (in surface water), 0.03 × 10 ⁵ –0.31 × 10 ⁵ items/km ² (in inflowing rivers), and 50–1292 items/m ² (in sediments)	Xiong et al., 2018
Yellow Sea, China	545 ± 282 items/m ³ (in surface seawater) and 37.1 ± 42.7 items/kg (in dried sediments)	Zhu et al., 2018
Northwestern Pacific	1.0 × 10 ⁴ items/km ²	Pan et al., 2018
Arctic Central Basin	0–375 items/m ³	Kanhai et al., 2018
Edgaston Pool, UK	250–300 items/kg (in dried sediments)	Vaughan et al., 2017
Atlantic Ocean	1.15 ± 1.45 items/m ³	Kanhai et al., 2017
Tuscany, Italy	0.26 items/m ³ (in the water-column) and 69,161.3 items/km ² (floating microplastics)	Baini et al., 2018
Balearic Islands, Spain	900,324 items/km ²	Ruiz-Orejón et al., 2018
Northwestern Mediterranean Sea	1.12 × 10 ⁵ items/km ²	Schmidt et al., 2018
Israeli Mediterranean coast waters	7.68 ± 2.38 items/m ³	Van der Hal et al., 2017
Xiangshan Bay, China	8.9 ± 4.7 items/m ³ (in surface seawater) and 1739 ± 2153 items/kg (in sediments)	Chen et al., 2018

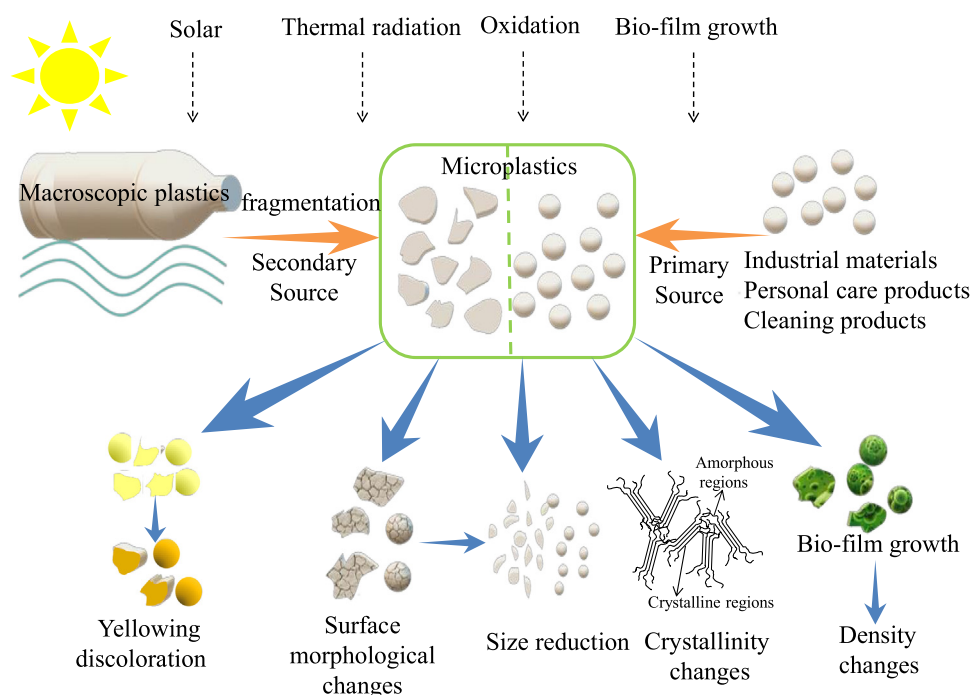


Fig. 1. Properties changes of microplastics after degradation.

Lambert and Wagner, 2016; Zhou et al., 2018; Zettler et al., 2013; Fazez and Ryan, 2016; Rouillon et al., 2016), as shown in Fig. 1.

3.1. Color changes

Laboratory study has reported that the color of ABS and PS particles turn white to yellow after one month of UV treatment (Kowalski et al., 2016). The plastic pellets collected from the Maltese Islands also exhibit different degree of yellowing discoloration, which may provide a qualitative measure of microplastics age (Turner and Holmes, 2011). When exposed to UV radiation source (such as sunlight), microplastics are prone to photo-degrade. The yellow reaction products, such as quinone compounds, are generated in this process. Thus, the color of microplastics shifts white to yellow. However, the photo degradation of microplastics in seawater is severely retarded, mainly because of the lower oxygen concentration and temperature (Andrady, 2011). In addition, the floating microplastics are more effective to develop surface fouling with biofilms than the land microplastics (Muthukumar et al., 2011; Andrady, 2011), which may keep microplastics sheltered from UV

radiation. Therefore, the yellowing discoloration rate of microplastics in marine environments was slower than in land.

3.2. Surface morphological changes

The surface morphological changes of microplastics are mainly attributed to the fragmentation by mechanical forces and the surface ablation by oxidative degradation. The mechanical forces such as hydraulic shear force and sand abrasion force could act on the surface of microplastics which consists of the amorphous component and the crystal component (Crawford and Quinn, 2017). The degradation rate of amorphous component is faster than the crystal component. Thus, cracks are generated in the surface and even in the deeper layers of the crystal component (Göpferich, 1996). The initial oxidative degradation generally takes place at the thin surface layer of microplastics, because oxygen cannot diffuse in the deep layer of microplastics (Gigault et al., 2016). Then UV radiation is attenuated within microplastics. This phenomenon results in the formation of minor cracks on the surface of microplastics. The SEM or the FE-SEM technology is frequently used to

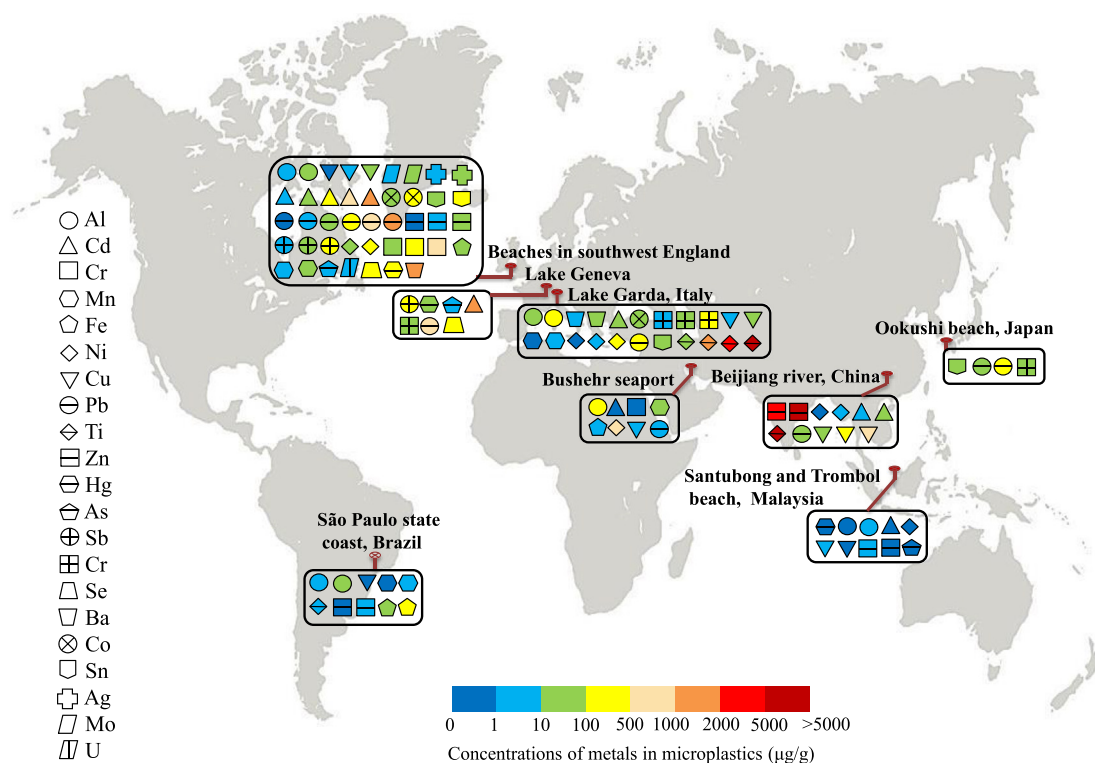


Fig. 2. Average concentration ranges of metals on microplastics.

characterize these surface morphological changes (Chen et al., 2018; Zhou et al., 2018; Sagawa et al., 2018). Microplastics samples collected from the Xiangshan Bay, China (Chen et al., 2018), the coastal beach in Bohai Sea and Yellow Sea, China (Zhou et al., 2018), the Hiroshima Bay, Japan (Sagawa et al., 2018), and the shorelines of the Great Lakes, America (Zbyszewski et al., 2014) are more rough compared with the virgin ones, the cracked textures are observed at the surface of microplastics by SEM or FE-SEM. Laboratory studies also show that after UV radiation, the minor microcrack, cracks and flakes are formed on the surface of the microplastics (Cai et al., 2018; Hüffer et al., 2018). Therefore, microplastics become somewhat embrittled, then they are broken into smaller plastics debris, even nanoplastics by further degradation processes (Lambert and Wagner, 2016; Gigault et al., 2016). Notably, the morphological changes and the reduction of particles sizes would increase the specific surface area of microplastics, and then affect the behaviors of microplastics in environments, such as the sorption behavior and the sinking behavior (Hüffer et al., 2018; Kowalski et al., 2016).

3.3. Crystallinity changes

The crystallinity of PP and PE increases through photo-oxidation or thermo-oxidation (Rouillon et al., 2016; Orden et al., 2015; Ter Halle et al., 2017). There are two reasons that contribute to explain this result. The first one is that the amorphous component in polymers is more inclined to degrade than the crystalline component in weathering/aging process. Thus, the fraction of crystalline component increased. The second one is that the chemi-crystallization in degradation process, which is caused by a random chain scission. The chemi-crystallization would increase the crystalline component of polymers (Fayolle et al., 2008; Rouillon et al., 2016). On the contrary, the crystallinity of copolymer reduces after degradation (Orden et al., 2015). The present of comonomer decreases the densities of polymers and facilitates oxygen into polymers (Ammala et al., 2011). Then the rupture of regularity domains occurs in degradation process, and the crystallization capabilities of copolymers deduce as a consequence (Orden et al., 2015).

3.4. Density changes

In aquatic environments, microplastics are inclined to be colonized by microorganisms, such as algae, bacteria and Fungi (Masó et al., 2003; Zettler et al., 2013; Johansen et al., 2018), and formed bio-film. Microplastics degradation by bio-film growth seems to play a minor role, because the majority of microplastics are resistance to biodegradation (Thompson et al., 2005). However, the growth of bio-film may play a major role in the distribution of microplastics in the water column, because it can increase the density of microplastics. We assume that the shape of microplastics is spherical with radius of R_0 (m), the density of microplastics is described as follows (Chubarenko et al., 2016):

$$\rho = \rho_0 + \rho_f \frac{3d}{R_0} \quad (1)$$

where ρ (g/L) is the average density of microplastics, ρ_0 (g/L) is the initial density of microplastics, ρ_f (g/L) is the density of biofilm ($\rho_f > \rho_0$), d (m) is the thickness of bio-film.

At the beginning of bio-film growth, d approached zero, microplastics particles have an average density of ρ_0 . Then d increases with time, which leads to an increase of the average density of microplastics. When microplastics are denser than water, they begin to sink in the water column. Microplastics could be even found in sediment samples (Dekiff et al., 2014).

4. Concentrations of pollutants on microplastics in global marine and coastal environments

Microplastics could accumulate pollutants, such as metals ions (Holmes et al., 2012; Hodson et al., 2017; Holmes et al., 2014; Brennecke et al., 2016), and organic pollutants (Fries and Zarfl, 2012; Bakir et al., 2014a; Li et al., 2018; Llorca et al., 2018; Wang and Wang, 2018a) from surrounding water because of their small size, huge specific area, and hydrophobic nature. The concentrations of pollutants on microplastics in global marine and coastal environments have been

studied by environmental monitoring (Ashton et al., 2010; Massos and Turner, 2017; Wang et al., 2017). Fig. 2 shows the average concentration ranges of metals on microplastics samples from environments, based on the literature data (Ashton et al., 2010; Nakashima et al., 2011; Holmes et al., 2012; Nakashima et al., 2012; Massos and Turner, 2017; Wang et al., 2017; Turner, 2016; Imhof et al., 2016; Vedolin et al., 2017; Noik et al., 2015; Filella and Turner, 2018; Dobaradaran et al., 2018) (Note that if microplastics are sampled at different sites in the same place, all of the average concentrations ranges are shown in Fig. 2). The average concentrations of metals on microplastics are distinct in different sampling locations. The concentrations of metals on microplastics from beaches of inland rivers/lakes in big cities (such as Beijiang River, China, Lake Garda, Italy, and Lake Geneva) are higher than coastal areas and remote areas (such as Santubong and Trombol in Kuching, Malaysia). For example, the concentrations of Zn on microplastics could reach up to 14,815 µg/g in sediment of Beijiang River, China (Wang et al., 2017), while this value is < 10 µg/g on microplastics from Santubong and Trombol in Kuching, Malaysia (Noik et al., 2015). However, at the same location, the concentrations of metals on microplastics may also distinct at different sampling time. Holmes et al. (2012) and Turner (2016) both studied the concentration of Pb on microplastics from beaches near the Plymouth in south west England. Holmes et al. (2012) reported that the maximum concentration of Pb was 8.82 µg/g. While Turner (2016) found that the maximum concentration of Pb was 17,500 µg/g (the media concentration was 87 µg/g), almost 4 orders of magnitude higher than the maximum concentration of Pb reported by Holmes et al. (2012). The concentrations of metals show temporal and spatial variability. The local factors, such as industrialization of the sampling locations and the presence of anthropogenic activities may affect the concentrations of metals on microplastics (Vedolin et al., 2017).

The average concentration ranges of organic pollutants (PAHs, PCBs, HCHs, and DDTs) on microplastics in global environments are depicted in Fig. 3, based on the literature data (Karapanagioti et al., 2011; Fisner et al., 2013a; Fisner et al., 2013b; Yeo et al., 2015; Fisner

et al., 2017; Mai et al., 2018; Taniguchi et al., 2016; Van et al., 2012; Zhang et al., 2015; Hosoda et al., 2014; Endo et al., 2005; Hirai et al., 2011; Heskett et al., 2012; Mizukawa et al., 2013; Frias et al., 2010; Antunes et al., 2013; Tang et al., 2018; Ogata et al., 2009). It is clear that the concentrations of PAHs on microplastics achieve high values (> 5000 ng/g) in East Asia, such as beaches near Bohai Sea, Yellow Sea, China and Ookushi, Japan. Microplastics from coast of São Paulo State, Brazil and southwest England also have high concentrations of PAHs. High concentrations of PCBs (> 2000 ng/g) on microplastics are found at coast São Paulo State, Brazil and beaches in Ookushi, Japan. In remote areas, such as Caribbean Sea, Hawallan islands and the east coast of Australia, the concentrations of PAHs and PCBs are much lower (< 500 ng/g). F. Wang et al. (2018) reviewed the PCBs, PAHs and HCHs on microplastics from big cities, small cities, and remote areas, suggested that the pollutants concentrations on microplastics from big cities were almost an order of magnitude higher than those from small cities and remote areas. In the case of HCHs and DDTs, the concentrations on microplastics are lower than PAHs and PCBs, especially in remote areas. For most studied areas, the concentrations of HCHs are < 10 ng/g. Therefore, Microplastics have different affinities for different kinds of pollutants.

The concentrations of pollutants on microplastics from environments are related to many factors, including the concentrations and types of pollutants, the environmental conditions in surrounding water, and the properties of microplastics (Vedolin et al., 2017; F. Wang et al., 2018). However, the environmental monitoring results only reveal the temporal and spatial distribution of the concentrations of pollutants on microplastics. It is difficult to get useful information about the effect of these factors on the accumulation or sorption of pollutants on microplastics because of the complexity of the real environments. On the contrary, the laboratory studies could investigate the factors by variable controlling approach. Thus, laboratory studies may provide useful ways to reveal the influencing factors and the possible mechanisms in the sorption process, which are overviewed in the following Sections.

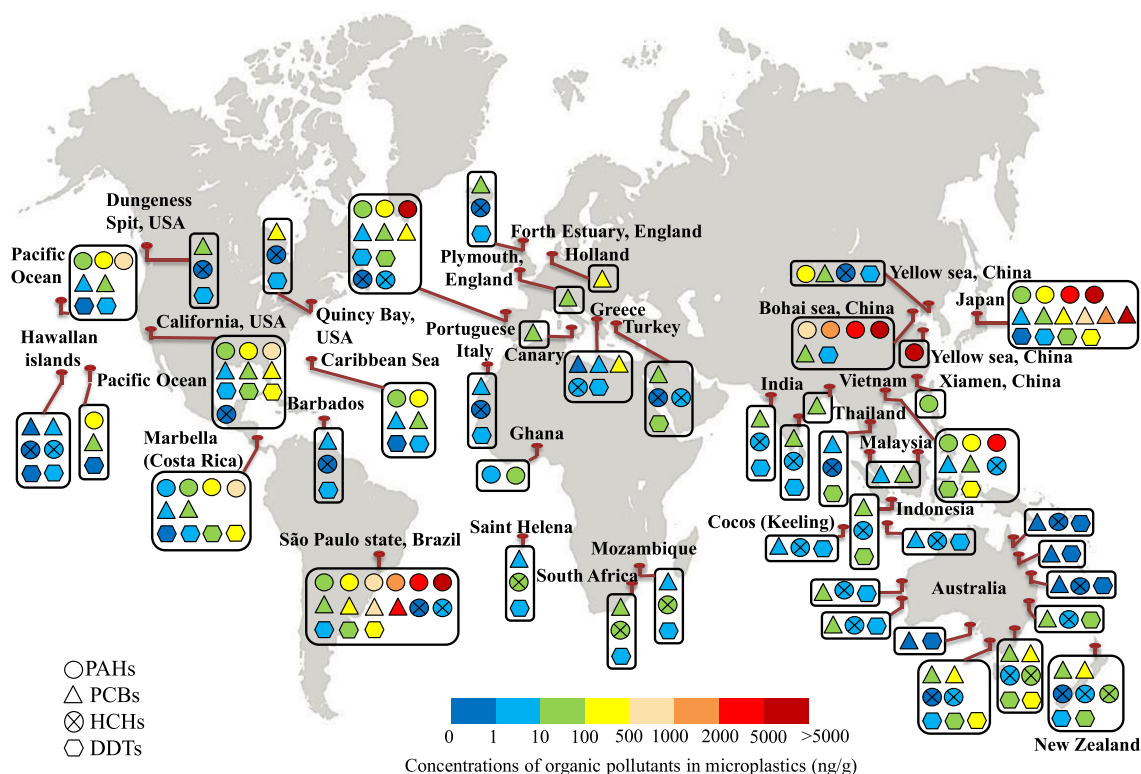


Fig. 3. Average concentration ranges of organic pollutants on microplastics.

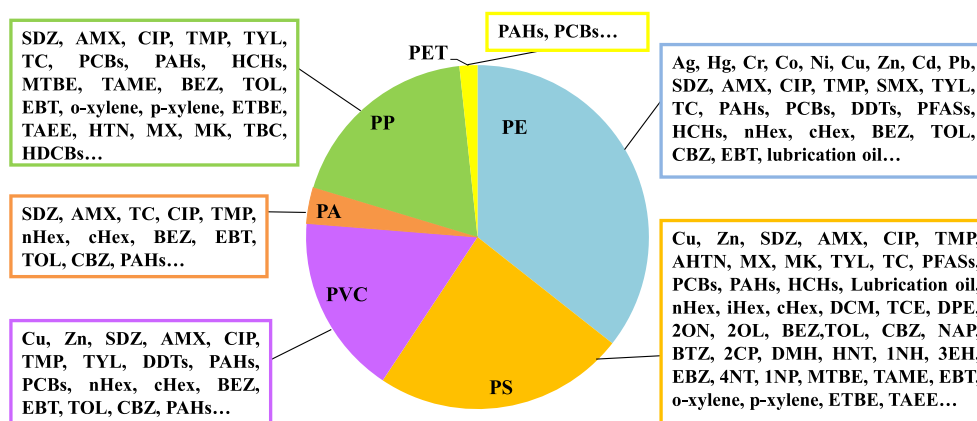


Fig. 4. Microplastics types and pollutants on microplastics (the size of each pie wedge represents the relative amount of the literature on the sorption behaviors of PE, PS, PVC, PA, and PP).

5. Sorption behaviors of microplastics and the influencing factors

5.1. Types and properties of microplastics

The most widely studied microplastics in laboratory are PE, PS, PVC, and PP, followed by PA and PET (Table 1). Fig. 4 summarizes those microplastics types and the pollutants which could be sorbed on microplastics, based on the literature data (Holmes et al., 2012; Li et al., 2018; Llorca et al., 2018; Wang and Wang, 2018a, 2018b; Bakir et al., 2012; Holmes et al., 2014; Brennecke et al., 2016; Bains et al., 2018; Hu et al., 2017; Liu et al., 2018a, 2018b; Hüffer et al., 2018; Z. Wang et al., 2018; Zhang et al., 2018; Guo et al., 2018; Fries and Zarfl, 2012; Lee et al., 2014; Hüffer and Hofmann, 2016; Müller et al., 2018; Xu et al., 2018a, 2018b; Bakir et al., 2014a; Rochman et al., 2013a, 2013b; Velzeboer et al., 2014; Karapanagioti et al., 2010; Karapanagioti and Klontza, 2008; Hodson et al., 2017; Zhan et al., 2016).

Different microplastics types have distinct sorption affinities for the same pollutants, which could be attributed to the differences in the rubbery domains, the polarity, and the functional groups of microplastics (Guo et al., 2012; Wang et al., 2015; Wang et al., 2015). The abundance of the rubbery domains of microplastics plays a major role in the sorption of organic compounds. The rubbery domains have high mobility, and high accessibility to some kinds of organic pollutants. (Guo et al., 2012). Under laboratory conditions, PE has abundance rubbery domains among microplastics (F. Wang et al., 2018). Researchers found that the sorption capacities of PYR, PHE, lubricating oils, PCBs, PFOS, POSA and 4, 4'-DDT on PE were higher than other types of microplastics (Bakir et al., 2012; Wang and Wang, 2018a, 2018b; Z. Wang et al., 2018; Rochman et al., 2013b; Wang et al., 2015). Müller et al. (2018) reported that PP with abundant rubbery domains showed high affinities for BTEX. O'Connor et al. (2016) demonstrated that the partitioning distribution of chemical substances on microplastics followed the order of LDPE \approx HDPE > PP > PVC > PS. In addition, the polarity and the special groups of microplastics could also influence the interactions between microplastics and polar compounds (Wang et al., 2015). For example, the polar polymer PA with amide group demonstrates higher sorption capacities for polar antibiotic CIP, TMP, AMX, and TC than other microplastics (Li et al., 2018).

Besides the types of microplastics, their properties (e. g., particles size and degree of aging) also play an important role in the sorption process. The sorption capacities of microplastics increase with decrease of particles size for the same types of microplastics, which is mainly due to the large specific surface areas of small particles (Zhan et al., 2016; Liu et al., 2018a, 2018b). In addition, the sorption equilibrium time of bigger particles is longer than smaller ones (F. Wang et al., 2018). For different types of microplastics, the specific surface areas may be different in the same particles size. Hüffer and Hofmann (2016) studied

the sorption of organic compounds on microplastics, found that the sorption was not followed the order of particles size. The effect of particles size is less important than the specific surface area and the microplastics types.

Assuming that all microplastics particles have the same size, the specific surface area can be described as follows:

$$S = \frac{S_{\text{total}}}{m_{\text{total}}} = \frac{nS_{\text{part}}}{m_{\text{total}}} = \frac{n(S_{\text{sur}} + S_{\text{pore}})}{m_{\text{total}}} \quad (2)$$

where S (m^2/g) is the specific surface area, S_{total} (m^2) and m_{total} (g) are the total surface area and the mass of all evaluated particles, n is the amount of the particles, S_{part} is the total surface area of one particle. S_{sur} (m^2) and S_{pore} (m^2) are the total area of the surface and the interior pores of one particle, respectively.

We can see from Eq. (2) that the specific area is related to the surface area and the pore area. Therefore, microplastics with rough surface, irregular shape and porous structure would have large specific surface area, and high sorption affinities for pollutants.

The weathering/aging process causes the degradation in microplastics, and therefore leads to the morphological changes, such as the formation of cracks and flakes on the surface of microplastics (Cai et al., 2018; Hüffer et al., 2018), as discussed in above Section. The specific surface areas of microplastics increase as a consequence of degradation. Evidence suggests that the aged microplastics had higher sorption affinities for pollutants than the virgin ones (Turner and Holmes, 2015; Brennecke et al., 2016; Müller et al., 2018). In marine and coastal environments, the aged microplastics could accumulate more pollutants than the virgin microplastics studied in laboratories, the concentrations of pollutants are even > 10,000 $\mu\text{g}/\text{g}$ in some areas (Turner, 2016).

5.2. Types of pollutants

5.2.1. Heavy metals

Heavy metals are very important pollutants which have aroused increasing attentions (Wang and Chen, 2009). A limited number of laboratory studies have investigated the sorption of metals on microplastics, as summarized in Table 3 (Holmes et al., 2012; Holmes et al., 2014; Turner and Holmes, 2015; Brennecke et al., 2016; Hodson et al., 2017). At low initial concentrations (< 20 $\mu\text{g}/\text{L}$), the uptakes of Ag, Cd, Co, Cr, Cu, Hg, Ni, Pb, and Zn by microplastics are ranging from 0.0004 to 2.78 $\mu\text{g}/\text{g}$ (Holmes et al., 2012; Holmes et al., 2014; Turner and Holmes, 2015). While at high initial concentrations of Zn (10^2 – 10^5 $\mu\text{g}/\text{L}$), the sorption capacities could reach up to 236–7171 $\mu\text{g}/\text{g}$ (Hodson et al., 2017). Brennecke et al. (2016) studied the sorption of Cu and Zn leached from an antifouling paint on microplastics in non-filtered seawater, and found that the maximum uptakes of Cu and Zn were about 3000 and 270 $\mu\text{g}/\text{g}$, respectively. These results demonstrate that the

Table 3
Summary of current studies of metals sorption on microplastics.

Metals	Microplastics	C ₀ (µg/L)	Environments	Experimental conditions	Sorption capacity (µg/g)	Reference
Ag	Virgin PE and beached pellets (PE > 90%)	2–20	River water, pH 4–10	48 h	0.0128–1.068 ^b	(Turner and Holmes, 2015)
Cd	Virgin and beached PE resin pellets	0–20	Filtered seawater, salinity 34‰, pH 7.8	7 days, 150 rpm, 20 ± 1 °C	0.0004–0.010 ^b	(Holmes et al., 2012)
	Virgin and beached PE pellets (< 1 mm)	0–20	Filtered river water and seawater (salinity 33.5 ‰), pH 4–10.5	48 h, 150 rpm	0.000429–0.248 ^b	(Holmes et al., 2014)
Co	Virgin PE and beached pellets (PE > 90%)	2–20	River water, pH 4–10	48 h	0.0101–0.248 ^b	(Turner and Holmes, 2015)
	Virgin and beached PE resin pellets	0–20	Filtered seawater, salinity 34‰, pH 7.8	7 days, 150 rpm, 20 ± 1 °C	0.018–0.038 ^b	(Holmes et al., 2012)
Cr	Virgin and beached PE pellets (< 1 mm)	0–20	Filtered river water and seawater (salinity 33.5 ‰), pH 4–10.5	48 h, 150 rpm	0.0176–0.0797 ^b	(Holmes et al., 2014)
	Virgin PE and beached pellets (PE > 90%)	2–20	River water, pH 4–10	48 h	0.0692–0.0796 ^b	(Turner and Holmes, 2015)
Cu	Virgin and beached PE resin pellets	0–20	Filtered seawater, salinity 34‰, pH 7.8	7 days, 150 rpm, 20 ± 1 °C	0.297–0.441 ^b	(Holmes et al., 2012)
	Virgin and beached PE pellets (< 1 mm)	0–20	Filtered river water and seawater (salinity 33.5 ‰), pH 4–10.5	48 h, 150 rpm	0.093–0.441 ^b	(Holmes et al., 2014)
Hg	Virgin PE and beached pellets (PE > 90%)	2–20	River water, pH 4–10	48 h	0.0933 ^b	(Turner and Holmes, 2015)
	Virgin PS beads (0.7–0.9 µm) and aged PVC fragments (1.6–0.8 mm)	–	Filtered seawater, salinity 34‰, pH 7.8	7 days, 150 rpm, 20 ± 1 °C	0.261 ^b	(Holmes et al., 2012)
Ni	Virgin and beached PE pellets (< 1 mm)	0–20	Filtered river water and seawater (salinity 33.5 ‰), pH 4–10.5	48 h, 150 rpm	0.101–0.263 ^b	(Holmes et al., 2014)
	Virgin PE and beached pellets (PE > 90%)	2–20	River water, pH 4–10	48 h	0.100 ^b	(Turner and Holmes, 2015)
Pb	Virgin PS beads (0.7–0.9 µm) and aged PVC fragments (1.6–0.8 mm)	–	Non-filtered seawater, water or no water exchange	14 days	0.09–3000 ^a	(Brennecke et al., 2016)
	Virgin PE and beached pellets (PE > 90%)	2–20	River water, pH 4–10	48 h	0.170–2.78 ^b	(Turner and Holmes, 2015)
Zn	Virgin PE and beached PE resin pellets	0–20	Filtered seawater, salinity 34‰, pH 7.8	7 days, 150 rpm, 20 ± 1 °C	0.008–0.070 ^b	(Holmes et al., 2012)
	Virgin and beached PE pellets (< 1 mm)	0–20	Filtered river water and seawater (salinity 33.5 ‰), pH 4–10.5	48 h, 150 rpm	0.00761–0.152 ^b	(Holmes et al., 2014)
Pb	Virgin PE and beached pellets (PE > 90%)	2–20	River water, pH 4–10	48 h	0.0166–0.152 ^b	(Turner and Holmes, 2015)
	Virgin and beached PE resin pellets	0–20	Filtered seawater, salinity 34‰, pH 7.8	7 days, 150 rpm, 20 ± 1 °C	0.716 ^b	(Holmes et al., 2012)
Zn	Virgin and beached PE pellets (< 1 mm)	0–20	Filtered river water and seawater (salinity 33.5 ‰), pH 4–10.5	48 h, 150 rpm	0.191–2.73 ^b	(Holmes et al., 2014)
	Virgin PE and beached pellets (PE > 90%)	2–20	River water, pH 4–10	48 h	0.191–2.74 ^b	(Turner and Holmes, 2015)
Zn	Virgin PE and beached pellets (PE > 90%)	2–20	River water, pH 4–10	48 h	0.1–0.7 ^b	(Turner and Holmes, 2015)
	Virgin PS beads (0.7–0.9 µm) and aged PVC fragments (1.6–0.8 mm)	–	Non-filtered seawater, water or no water exchange	14 days	0.18–270 ^a	(Brennecke et al., 2016)
	Plastic bag derived HDPE (0.92 ± 1.09 mm ²)	10 ² –10 ⁵	Background electrolyte (0.1 M NaNO ₃)	48 h, 220 rpm	236–7171	(Hodson et al., 2017)

^a Ranges of sorption uptakes estimated from figures in literature.

^b Sorption capacities calculated from the Langmuir model.

concentrations of metals in surrounding water play a major role in the sorption processes. This result could help to explain that the concentrations of metals on microplastics from marine and coastal environments vary from 10^{-1} to 10^4 $\mu\text{g/g}$ at different locations, as mentioned in above Section.

Efforts have been made to study the mechanisms of metals sorption on microplastics. For most studies, the sorption isotherm and kinetics models are used to model the experimental data and to investigate the mechanisms in the sorption process. The sorption isotherm of metals on microplastics can be represented by the Freundlich isotherm (Hodson et al., 2017), which indicate that the multilayer sorption occurs. Holmes et al. (2012) and Holmes et al. (2014) found that the both the Freundlich and the Langmuir isotherms described adequately the sorption of metals on microplastics. Therefore, the monolayer sorption on active sites of microplastics may also occur. In addition, the sorption kinetics of metals on microplastics follows the PFO model (Holmes et al., 2012). However, those sorption models are almost empirical models, and lack of specific physical meanings. The sorption process has three steps: 1, the diffusion of metals in the films around the microplastics; 2, the diffusion of metals inside the pore in microplastics, and 3, the adsorption on the active sites. The parameters of sorption phenomenon models (e.g. the mass transfer models and the pore-film diffusion models) have clear physical meanings and can describe the overall sorption steps (Marin et al., 2014; Ocampo-Perez et al., 2017; Suzaki et al., 2017). Until recently, scarce amount of literature has studied the sorption phenomenon models of metals sorption on microplastics, which are highly recommended in the future work.

5.2.2. Organic pollutants

Organic pollutants such as antibiotics are ubiquitous in aquatic environments (Wang et al., 2019a, 2019b; Liu et al., 2018a, 2018b; Wang and Wang, 2016). The sorption of PAHs, PCBs, HCHs, DDTs, antibiotics, PFASs, flame retardants, fuel aromatics, ethers, SM, and other organic pollutants on microplastics are summarized in Table 4. Organic pollutants with different polarity and hydrophobicity show different affinities to microplastics. Hüffer and Hofmann (2016) suggested that the sorption capacities of nonpolar organic pollutants (i.e. nHex, cHex, BEZ, TOL, CBZ, EBT, and NAP) were higher on nonpolar polymers than polar polymers. The polar organic pollutants may have high affinities for polar polymers, but the polarity of polymers is not the only influencing factor, the special groups of polymers are also significant factors that influence the sorption of polar organic pollutants (Li et al., 2018).

Organic pollutants have different hydrophobicity. In general, the hydrophobicity of organic pollutants follows the order of HCHs < PCBs < DDTs < PAHs (F. Wang et al., 2018). Evidence suggests that the partition coefficients of organic pollutants on microplastics are strongly related to the hydrophobicity (Ziccardi et al., 2016; Hartmann et al., 2017; F. Wang et al., 2018). As shown in Table 2, the uptakes of PCBs, and PAHs in laboratory study are generally higher than DDTs (Bakir et al., 2012; Wang and Wang, 2018a, 2018b; Z. Wang et al., 2018; Bakir et al., 2014a; Bakir et al., 2014b; Zhan et al., 2016; Hüffer et al., 2018; Hüffer and Hofmann, 2016). Environmental monitoring results (Fig. 3) also demonstrate that the concentrations of PAHs and PCBs on microplastics are higher than DDTs and HCHs on microplastics.

There have been many attempts to studies the sorption mechanisms of organic pollutants on microplastics. Evidence shows that the sorption equilibrium mainly followed the Langmuir and the linear models (Hu et al., 2017; Wang and Wang, 2018a, 2018b; Xu et al., 2018a, 2018b; Zhan et al., 2016). Thus, the monolayer sorption on homogeneous surface of microplastics or the hydrophobic interaction is the main mechanisms. In addition, the Freundlich model could also describe some sorption processes where the multilayer sorption may occur (Guo et al., 2012; Hüffer and Hofmann, 2016; Xu et al., 2018a, 2018b). The PFO, the PSO and the intra-particle models are frequently applied to

model the sorption kinetics of organic pollutants on microplastics (Xu et al., 2018a, 2018b; Wang and Wang, 2018a, 2018b; Llorca et al., 2018). However, the lack of parameters having a physical meaning of the PFO and PSO models narrowed the meanings of investigating the sorption mechanisms by models, as mentioned in above Section. The parameters of the intra-particle model have a physical meaning, but can only describe the pore diffusion step of the sorption process. The film-diffusion and adsorption on active sites have not been modeled in the sorption of pollutants on microplastics. Therefore, the sorption phenomenon models are needed in future studies.

5.3. Experimental environments

The effect of salinity of the surrounding water on the sorption of pollutants on microplastics was related to the mechanisms involved in the sorption process (Crawford and Quinn, 2017). The surface of microplastics is negatively charged in solution (Li et al., 2018), and the presence of salt increase the concentration of Na^+ . The positively charged Na^+ was inclined to interact with microplastics through electrostatic attraction. Therefore, if the electrostatic interaction was the main mechanism in the sorption of pollutants on microplastics, Na^+ may compete with the pollutants ions, and decreased the sorption capacities of pollutants ions on microplastics. For example, Holmes et al. (2014) demonstrated that the sorption capacities of metals on beached pellets were higher in river water than in seawater. Llorca et al. (2018) suggested that the presence of salt decrease the sorption of PFASs on microplastics. The increase of salinity also decreases the partition coefficient of α -HCHs on microplastics (F. Wang et al., 2018). If the electrostatic interaction was not the main mechanism, the salinity may have minor influence on the sorption process. For example, the changes in salinity have no significant influence on the sorption of TC (Xu et al., 2018a, 2018b) and SM (Zhang et al., 2018) on microplastics. The presence of salinity may also increase the sorption capacities of pollutants on microplastics, probably due to the sensitivity of outer-sphere surface complexation to ionic strength (Hu et al., 2017). The sorption capacities of lubricating oil on PE and PS remarkably increase with increase of salinity (Hu et al., 2017). Similar conclusion was reported by Wang et al. (2015); they found that increase of salinity would increase the affinities of PE for PFOS. The presence of salt increases the partition coefficient of most PAHs on microplastics (F. Wang et al., 2018). However, until present, the clear mechanisms involved have not been elucidated. The molecular simulation and density functional theory study which could reveal the sorption mechanisms are recommended in the future work.

Theoretically, pH could affect the electro negativity of microplastics polymers, and further affect the sorption behaviors (Xu et al., 2018b). Microplastics are always negatively charged in alkaline solution. And the surface of polymers is protonated or positively charged in acid solution (Wang et al., 2015). Therefore, alkaline environments would conduce to the sorption of cations on microplastics, due to the electrostatic attraction. For example, Holmes et al. (2014) found that the sorption of Cd, Co, Ni, and Pb on virgin and beached PE pellets increased with increase of pH (the sorption of Pb increased to a certain pH value and then relatively constant with pH). On the contrary, alkaline environments are unfavorable for the sorption of negatively charged ions on microplastics because of the electrostatic repulsion. For example, Wang et al. (2015) reported that the sorption capacity of PFOS on PE decreased with increase of pH. Moreover, the changes in pH values have minor effect on the sorption when the interactions of non-ionic forms and the surface of polymers occur. Laboratory evidence shows that lubricating oil exhibits no clear dependence on pH (Hu et al., 2017).

6. Impact of microplastics to marine organisms

Microplastics in environments could be ingested by marine

Table 4
Summary of current studies of organic pollutants sorption on microplastics.

Types	Organic pollutants	Microplastics	C ₀	Environments	Experimental conditions	Sorption capacity (µg/g)	Reference	
PAHs	PHE	PVC and PE (200–250 µm)	0.8–3.1 µg/L	Filtered seawater	24 h, 220 rpm, 20 °C	1.15–15.5	(Bakir et al., 2012)	
		PE, PS, and PVC (100–150 µm)	10–200 µg/L	Artificial freshwater (AFW)	120 h, 200 rpm, 20 °C	303.03–714.29 ^b	(Wang and Wang, 2018a)	
		PE and nylon (0.2–2 mm)	100–1000 µg/L	Synthetic seawater, salinity 25‰	48 h, 150 rpm, 25.0 °C	270–310 ^a	(Z. Wang et al., 2018)	
		PVC and PE (200–250 µm)	0.2–10 mg/L	Seawater, salinity 0–25‰	24 h, 220 rpm, 18 °C	2–4	(Bakir et al., 2014a)	
PCBs	27 PCBs	PE pellets and PP	0.6–6.1 µg/L	Synthetic seawater (SW)	150 days, shaken periodically, 23 °C	–	(Karapanagioti et al., 2010)	
		PP (2 mm), PE (2–3 mm) and POM (2–3 mm)	100 µg/L	Synthetic freshwater (FW) and synthetic saltwater (SW)	164 days, shaken periodically, 23 ± 2 °C	–	(Karapanagioti and Klontza, 2008)	
		LDPE (4.2 × 4.7 × 2.8 mm) and HDPE (4.0 × 4.4 × 2.0 mm)	1–100 ng/L	Aqueous solutions	7 days, 300 rpm, 20 ± 1 °C	–	(Fries and Zarfl, 2012)	
		PE, PP and PS (< 250 µm)	–	Artificial seawater	150 rpm, 25 °C	–	(Lee et al., 2014)	
DDTs	4, 4' -DDT	PS pellets (3 × 2 mm)	–	San Diego Harbor Excursions and Shelter Island	1, 3, 6, 9, and 12 months field measurement	0.063–0.208	(Rochman et al., 2013 a)	
		PE, PS, and PVC (100–150 µm)	0–100 µg/L	Artificial freshwater (AFW)	120 h, 200 rpm, 25 °C	78.7–333 ^b	(Wang and Wang, 2018a, 2018b)	
		PE pellets and PP	2 µg/L	Synthetic seawater (SW)	150 days, 23 °C	–	(Karapanagioti et al., 2010)	
		PE, PP and PS (< 250 µm)	–	Artificial seawater	150 rpm, 25 °C	–	(Lee et al., 2014)	
HCHs	α-HCH β-HCH γ-HCH δ-HCH	PET pellets (3 × 2 mm)	300 µg/L	Five locations throughout San Diego Bay	12-months field measurement	0.0136–0.203	(Rochman et al., 2013 a)	
		PE pellets and PP	–	Synthetic seawater (SW)	150 days, shaken periodically, 23 °C	–	(Karapanagioti et al., 2010)	
		Aged PS (125–250 µm)	–	Aqueous solution. (0.01 M CaCl ₂)	7 days, 125 rpm, 25 ± 2 °C	Within 10 ⁻² –10 ^{1a}	(Hüffer et al., 2018)	
		PE, PA, PS and PVC (< 250 µm)	Range over 3 or 4 orders of magnitude	Aqueous solution (10 mM CaCl ₂ and 3.08 mM Na ₂ S)	17 days (PS), 5 days (PE, PA, PVC), 10 rpm, 25 °C	Within 1 ⁻¹⁰ –10 ^{2a}	(Hüffer and Hofmann, 2016)	
PCBs	ANT, CHR, BKF CHR-d12, FLO, BaP, CHR, TRI, BbF	LDPE (4.2 × 4.7 × 2.8 mm) and HDPE (4.0 × 4.4 × 2.0 mm)	1–100 ng/L	Aqueous solutions	7 days, 300 rpm, 20 ± 1 °C	–	(Fries and Zarfl, 2012)	
		PET pellets (3 × 2 mm)	–	San Diego Harbor Excursions and Shelter Island	1, 3, 6, 9, and 12 months field measurement	0–0.126	(Rochman et al., 2013 a)	
		PET cylinders (3 × 2 mm) HDPE, PVC, LDPE and PP (3 mm)	–	Five locations throughout San Diego Bay	12-months field measurement	0.002–0.034	(Rochman et al., 2013 b)	
		PP (0.18–5 mm)	0.05–2.5 mg/L	Ultrapure water, simulated seawater, and n-hexane	24 h, 220 rpm, 25 °C	93.45–344.8 ^b	(Zhan et al., 2016)	
DDTs	4, 4' -DDT	PE (10–180 µm) and PS (70 mm)	Within 10 ⁻⁵ –10 ⁻¹ µg/L	Fresh and seawater (salinity 34‰)	6 weeks, 100 rpm, 20 °C	–	(Velzeboer et al., 2014)	
		PE, PP and PS (< 250 µm)	650 µg/L 100 µg/L 600 µg/L 600 µg/L	Artificial seawater	150 rpm, 25 ± 2 °C	–	(Lee et al., 2014)	
		PVC and PE (200–250 µm)	0.8–3.1 µg/L	Filtered seawater	24 h, 220 rpm, 20 °C	1.32–6.18	(Bakir et al., 2012)	
		PVC and PE (200–250 µm)	0.6–6.1 µg/L 0.8–3.1 µg/L 0.8–3.1 µg/L	Seawater, salinity 0–35‰	24 h, 220 rpm, 18 °C	1.5–10 ^a	(Bakir et al., 2014a)	
DDTs	4, 4' -DDT	PVC and PE (200–250 µm)	0.8–3.1 µg/L	Seawater	24–360 h, 220 rpm, 18 °C	1.2–6.2 ^b	(Bakir et al., 2014b)	

(continued on next page)

Table 4 (continued)

Types	Organic pollutants	Microplastics	C ₀	Environments	Experimental conditions	Sorption capacity (µg/g)	Reference
Antibiotics	SDZ	PE, PS, PP, PVC, and PA, (75–180 µm)	0.5–15 mg/L	Ultrapure water and filtered seawater	4 days, 180 rpm, 25 °C	0–100 ^a	(Li et al., 2018)
	AMX					0–3000 ^a	
	CIP					0–1000 ^a	
	TMP					40–100 ^a	
PFASs	SMX	PE (150 µm)	0.2–5 mg/L	Aqueous solution, pH 2–12,	72 h, 200 rpm, 25 °C	25–700 ^b	(Xu et al., 2018b)
	TYL	PE, PP, PS, and PVC	1–30 ppm	Aqueous solution	48 h, 150 rpm, 25 °C	1667–3333 ^b	(Guo et al., 2018)
	TC	PE, PP, and PS (< 280 µm)	0.2–5 mg/L	Aqueous solution, pH 2–12, salinity 0.5‰–35‰	24 h, 200 rpm, 25 °C	109–167	(Xu et al., 2018a)
		PA, (75–180 µm)	0.5–15 mg/L	Ultrapure water and filtered seawater	4 days, 180 rpm, 25 °C	50–2000 ^b	(Li et al., 2018)
Flame retardants	PFOS	PE (150 µm), PS (250 µm), and PVC (130 µm)	5–50 µg/L	Aqueous solutions, pH 3–7	7 days, 150 rpm, 25 °C	0–1.4 ^b	(Wang et al., 2015)
	FOSA	PE (150 µm), PS (250 µm), and PVC (130 µm)	5–50 µg/L	Aqueous solutions, pH 3–7	7 days, 150 rpm, 25 °C	1.4–2.4 ^b	(Wang et al., 2015)
	PFASs	HDPE (3–16 µm), PS(10 µm), and PS-COOH (10 µm)	1–20 µg/L	Fresh water and seawater	50 days, 120 rpm, 20 °C	34–210 (Σ PFASs) ^b	(Llorca et al., 2018)
		PP (0.45–0.85 mm)	10 µg/L	Simulated seawater, salinity 35‰	24 h, 220 rpm, 18 °C	21.489 ^b	(Liu et al., 2018a, 2018b)
Fuel aromatics	HBCDs	Virgin and aged PS (3.5 × 2.2 mm) and PP (3–5 mm)	2 µg/L	Tab water	Two weeks, 10 rpm, 22 °C	1.954–3.34 ^b	(Müller et al., 2018)
	BEZ	Aged PS (125–250 µm)	130–190 µg/L			–	
	TOL	PE, PA, PS and PVC (< 250 µm)	–	Aqueous solution. (0.01 M CaCl ₂)	7 days, 125 rpm, 25 ± 2 °C.	Within 10 ⁻² –10 ^{4a}	(Hüffer et al., 2018)
		Virgin and aged PS (3.5 × 2.2 mm) and PP (3–5 mm)	Range over 3 or 4 orders of magnitude	Artificial seawater	17 days for PS, 5 days for PE, PA, and PVC, 10 rpm, 25 °C	Within 1–10 ^{4b}	(Hüffer and Hofmann, 2016)
Ethers	EBT	Aged PS (125–250 µm)	130–190 µg/L	Tab water	Two weeks, 10 rpm, 22 °C	–	(Müller et al., 2018)
	EBM	PE, PA, PS and PVC (< 250 µm)	–	Aqueous solution. (0.01 M CaCl ₂)	7 days, 125 rpm, 25 ± 2 °C.	Within 10 ⁻² –10 ^{4a}	(Hüffer et al., 2018)
	PX	PE, PA, PS and PVC (< 250 µm)	Range over 3 or 4 orders of magnitude	Artificial seawater	17 days for PS, 5 days for PE, PA, and PVC, 10 rpm, 25 °C	Within 0.1–10 ^{3a}	(Hüffer and Hofmann, 2016)
	OX	PE, PA, PS and PVC (< 250 µm)	Range over 3 or 4 orders of magnitude	Artificial seawater	17 days for PS, 5 days for PE, PA, and PVC, 10 rpm, 25 °C	Within 10–10 ^{4a}	(Hüffer and Hofmann, 2016)
Other organic compounds	CEZ	Virgin and aged PS (3.5 × 2.2 mm), and PP (3–5 mm)	130–190 µg/L	Tab water	Two weeks, 10 rpm, 22 °C	–	(Müller et al., 2018)
	MTBE	PE, PA, PS and PVC (< 250 µm)	Range over 3 or 4 orders of magnitude	Artificial seawater	17 days for PS, 5 days for PE, PA, and PVC, 10 rpm, 25 °C	Within 1–10 ^{4b}	(Hüffer and Hofmann, 2016)
	TAME	Virgin and aged PS (3.5 × 2.2 mm) and PP (3–5 mm)	130–190 µg/L	Tab water	Two weeks, 10 rpm, 22 °C	–	(Müller et al., 2018)
	ETBE	PP (0.125–5 mm)	5 µg/L	Simulated seawater, salinity 35‰	24 h, 220 rpm (for 30 min), room temperature	1.3	(Zhang et al., 2018)
nHex	TAE	PE (20–140 µm) and PS (50 µm)	Range over 1 or 2 orders of magnitude.	Aqueous solution, NaCl 0.001 to 0.1 mol/L, pH 1–10,	48 h, 293 K	6,800,000 ^b	(Hu et al., 2017)
	AHTN	PE, PA, PS and PVC (< 250 µm)	Range over 3 or 4 orders of magnitude	Artificial seawater	17 days for PS, 5 days for PE, PA, and PVC, 10 rpm, 25 °C	5,200,000 ^b	(Hüffer and Hofmann, 2016)
	MX						
	MK						

^a Sorption ranges estimated from figures in literature.

^b Sorption capacities calculated from the Langmuir model.

organisms, such as zooplankton (Cole et al., 2013; Desforges et al., 2015; Rje et al., 2017), mussels (Cole et al., 2015; Kolandhasamy et al., 2018; Qu et al., 2018), oysters (Leslie et al., 2017), corals (Hall et al., 2015), fish (Collard et al., 2015; Hipfner et al., 2018), turtles (Caron et al., 2018), and even seabirds (Provencher et al., 2018). Microplastics pose risks to marine organism once they are ingested by these organisms.

Physically, microplastics lead to the mechanical damages to organisms. For example, microplastics could cause the blockage of the digestive tract, the intestinal damage (including the cracking of villi and the splitting of enterocytes), and even alter the filtering activity and phagocytosis of organisms (Lei et al., 2018; Wegner et al., 2012; Canesi et al., 2015). These damages may finally lead to the death of organisms. Moreover, microplastics could accumulate in food web by predation. For example, Setälä et al. (2014) found that microplastics were fed through pelagic food web (from zooplankton to mysid shrimps). Cedervall et al. (2012) found that microplastics transferred from algae to zooplankton to goldfish. Thus, microplastics may accumulate in organisms in several alternate routes. Microplastics in seafood as protein sources may also cause potential health risks to human.

Chemically, microplastics in aqueous environments would sorb and accumulate pollutants, as mentioned in above Sections. The ingestion of contaminated microplastics may therefore introduce toxic pollutants to marine organisms, and even to food web (Bakir et al., 2014b; Brennecke et al., 2016; Santana et al., 2017). In this sense, microplastics act as vectors for toxic pollutants. But there are still no clear evidence for the effect of trophic transfer of microplastics and contaminants from food web to human health (Carbery et al., 2018), which should be further studied. Koelmans et al. (2016) suggested that the fraction of total HOCs accumulated on microplastics was small compared with other media in the marine environments, and the ingestion of microplastics by marine organisms may not increase the risks of HOCs. However, Wang and Wang (2018b) reported that the sorption capacities of PHE on PE, PS and PVC microplastics were higher than the sorption capacities on sediment samples. In addition, the concentrations of Ni, Cd and Zn on microplastics from Beiji River, China are 1–3 orders of magnitudes greater than the concentrations in sediment (Wang et al., 2017). These results suggest that in some cases, microplastics could accumulate more pollutants than other media in environments, and they are still important carriers of pollutants to marine organisms.

Biologically, microplastics are inclined to be colonized by microorganisms. Microplastics could influence the evolution of microbial communities and the exchange of genes (such as antibiotic resistance genes and metal resistance genes) between different bacterial taxa (Yang et al., 2019). Yang et al. (2019) suggested that the microbial community composition was the determining factor of the antibiotic resistance genes profile.

7. Conclusions and recommendations for further work

This review analyzes the accumulations and sorption of organic pollutants and metals on microplastics and the potential influencing factors. The concentrations of pollutants on microplastics in marine and coastal environments show spatial differences. Concentrations of pollutants on microplastics from remote areas are lower than that in big cities. The concentrations of HCHs and DDTs on microplastics are lower than PAHs and PCBs in global environments. In addition, the types and properties of microplastics, the types and concentrations of pollutants, and the experimental conditions (salinity and pH) could affect the sorption behaviors of microplastics. Moreover, microplastics in environments undergo weathering/aging processes, which cause the degradation of microplastics. The degradation mainly changes the color, surface morphological, particles size, crystallinity, and densities of microplastics. The aged microplastics have higher sorption capacities of pollutants than the virgin ones. Furthermore, the ingestion of contaminated microplastics by marine organisms may lead to the

introduction of contamination to the marine organisms and food web and bring risks to marine organisms.

The future works are recommended as follows:

1. To establish the standard methods for sampling and analyzing of microplastics in marine and coastal environments. The abundance, distribution and properties of microplastics in global environments need statistical monitoring and assessment based on the standard methods or criterion.
2. To relevant the laboratory studies and environmental application. Long term field sorption experiments are recommended in the future work.
3. To study the mechanisms of pollutants sorption on microplastics, and to evaluate the risk and fate of microplastics in environments.
4. To further study the interactions of microplastics and wider ranges pollutants, (e.g. radioactive heavy metals and antibiotics).
5. To evaluate the risk of microplastics to marine organisms and human. Studies to evaluate the microplastics as vectors to transport pollutants, especially toxic pollutants through food webs are needed in this sense.

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