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Review

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# Microplastics in aquatic environments: occurrence, accumulation, and biological effects

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**ABSTRACT**

Microplastics, whether originating directly from industrial and household products or from the degradation of larger plastics, are currently of intense global concern. These particles are present in aquatic environments in high concentrations and may adversely affect aquatic organisms. An additional concern is the ability of microplastics to adsorb inorganic and organic pollutants and subsequently liberate them into marine and freshwater systems. In this review, we report on the occurrence and abundance of microplastics in the global aquatic environment. We then consider the accumulation (uptake, distribution, and elimination) of microplastics in aquatic organisms and the important factors that lead to bioaccumulation. The effects of microplastics on aquatic organisms of different trophic levels are also discussed. Several studies have shown that the size, shape, and surface physicochemical characteristics of microplastics are essential determinants of their biological effects. Finally, we examine the combined effects of microplastics and other pollutants, including persistent organic pollutants and heavy metals. Our review concludes by suggesting future lines of research based on the remaining knowledge gaps in microplastic research.

*Keywords:* Bioaccumulation; Combined effects; Environmental microplastic concentration; Toxicity

## 1. Introduction

The material characteristics of plastics, especially their light weight, durability, and corrosion resistance, account for their widespread use in industry and in daily life. The plastic industry has grown enormously since 1950 such that global plastics production reached 348 million tons in 2017 (Plastics Europe, 2018) and will likely reach 33 billion tons by 2050 (Rochman et al., 2013). However, the extensive use, high-level production, rapid disposal, and recalcitrance to degradation of plastics have resulted in their large-scale release into aquatic and terrestrial environments, where they will persist for centuries. Rivers transport 70–80% of plastics, leading to their extensive deposition in the world's oceans (Horton et al., 2017). As one of the fastest-growing sources of pollution, plastic debris has become an environmental concern of high priority. Nonetheless, while plastic debris ranges in size from microscopic particles to pieces meters in size, it is microplastics that are currently the causes of great public concern.

Microplastics are generally defined as synthetic polymer particles < 5 mm in diameter (Morre, 2008, Zarfl et al., 2011, von Moos et al, 2012). They can originate from either primary or secondary sources. Primary sources include textiles, medicines, and personal care products such as facial and body scrubs (Rochman et al., 2015, Hernandez et al., 2017) as well as the manufactured pellets used for feedstock and plastic production (Lechner et al., 2014). Secondary sources derive from the breakdown of larger plastic litter items, including plastic containers, nets, line fibers, films, and tires (Alimi et al., 2018), via weathering and aging processes, such as UV radiation, wave and wind abrasion, hydrolysis, thermal

degradation, and microbial degradation. Primary microplastics are commonly released into domestic and industrial wastewater, finally entering rivers and estuaries (Hidalgo-Ruz et al., 2012). However, secondary microplastics are considered to be the largest source of microplastic pollution in aquatic environments (Eriksen et al., 2013; Hidalgo-Ruz et al., 2012) and their abundance will increase enormously with the continuous input of plastic debris from different origins. Moreover, because of their small size and large specific surface area, microplastics are bioavailable to a wide range of aquatic organisms and may therefore induce toxic effects whose impacts may spread throughout the food chain. Among the aquatic organisms known to ingest microplastics are amphipods, copepods, lugworms, barnacles, mussels, decapod crustaceans, seabirds, fish, and turtles (Desforges et al., 2015; Nelms et al., 2018; Alomar et al., 2017; Van Cauwenberghe et al., 2015; Abbasi et al., 2018; van Franeker et al., 2011; Caron et al., 2018). Toxic effects on growth, reproduction, and survival have been demonstrated, together with oxidative stress and neurotoxicity (Yu et al., 2018; Tang et al., 2018; Wang et al., 2019c; Brandts et al., 2018a; Blarer et al., 2016).

In contrast to the considerable research on the abundance and distribution of microplastics in aquatic ecosystems, much remains to be learned about the bioaccumulation and biological effects of microplastics in marine and freshwater environments. In this review, we discuss recent research findings on the occurrence, bioaccumulation, and toxicity of microplastics in aquatic ecosystems. We begin by providing estimates of the abundance of microplastics. We then review existing studies on the accumulation of microplastics in aquatic organisms and the potential risks of microplastics in aquatic ecosystems, including

combined effects with chemical pollutants. Our review concludes by offering suggestions for future directions of microplastic research.

## 2. Microplastics in aquatic environments

### *2.1. Microplastics in marine systems*

A large amount of plastic waste flows into the ocean each year. In the Black Sea, plastic litter inputs via the Danube are estimated to be ~7.5 mg/m$^3$/s, corresponding to an annual average of 1553 tons (Lechner et al., 2014). By 2025, 250 Mt of plastics will have accumulated in the ocean (Wright et al., 2017). Due to the actions of wind, waves, microbial degradation, and UV radiation, large plastic debris in the ocean will eventually break down into macroplastics (> 25 mm in size), mesoplastics (5 – 25 mm), and microplastics (< 5 mm) (Alimi et al., 2018). Further substantial plastic inputs are microplastics from industrial and household products, which are released into sewage systems and eventually flow into the sea.

The presence of microplastics has been reported in marine systems worldwide (Auta et al., 2017; Zhao et al., 2015; Wang et al., 2019a; Pan et al., 2019) (Table 1). For example, the surface waters of the Northwestern Pacific Ocean are extensively polluted by microplastics, with concentrations ranging from 640 to 42000 items/km$^2$ depending on the action of currents (Pan et al., 2019). In the surface waters of the Arabian Bay, microplastic concentrations are in the range of $4.38\times10^4$ – $1.46\times10^6$ items/km$^2$ (Abayomi et al., 2017). Microplastics are also present in abundance in the semi-enclosed ocean Mediterranean, which receives large amounts of plastic debris from the surrounding land masses (Collignon et al., 2012). Generally, microplastic levels are much higher in waters adjacent to highly urbanized areas

than to rural areas. In a study sampling the seawater off the coast of South Korea, the mean microplastic abundance in urban coastal areas was as high as 1051 particles/$m^3$, compared to 560 particles/$m^3$ in rural coastal areas (Song et al., 2018). Microplastics have even been found in polar regions. In surface and sub-surface Arctic water samples collected south and southwest of Svalbard, Norway, microplastic concentrations of 0 – 1.31 and 0 – 11.5 items/$m^3$, respectively, were determined (Lusher et al., 2015). In the sub-water off Northeast Greenland, the microplastic concentration ranges from 1 to 3 items/$m^3$ (Morgana et al., 2018), and in the vicinity of the Antarctic Peninsula (latitudes of 61°S and 64°S) from 755 to 3524 items/$km^2$ (Lacerda et al., 2019). The most commonly detected polymer types are polypropylene (PP), polyethylene (PE), polyvinylchloride (PVC), polystyrene (PS), and polytetrafluoroethylene (PTFE).

Sediments are the final destination of microplastics in marine environments and microplastics are present ubiquitously in marine sediments (Woodall et al., 2014) (Table 1). A recent report showed that the average concentration of microplastics in the offshore sediments of the Yellow Sea and East China Sea is 155 and 142 items/kg dry weight (dw), respectively (Zhang et al., 2019b). In a study of the surface sediments of the northern Bering Sea and the Chukchi Sea, the abundance of microplastics ranged from below the detection limit to 68.9 items/kg dw (Mu et al., 2018). Water flow rate, sediment depth, and distance from the shoreline affect the abundance of microplastics in marine sediments. In the Maowei Sea, the microplastic concentration in the sediments of the estuary (520–940 items/kg dw) are much lower than at the entrance zone (1780 – 2310 items/kg dw) (Li et al., 2019). The

authors of that study suggested that lower water flow rates at the entrance accelerate the deposition of microplastics from water to the sediment. The average abundance of microplastics in the sediments of the South Yellow Sea was shown to positively correlate with sampling depth. In a recent study, the concentration of microplastics in sediments at depths < 20 m, 20 – 40 m, 40 – 60 m, and 60 – 80 m was 1765, 2135, 2346.7, and 2771.3 items/kg dw, respectively, indicating greater accumulation by deeper sediments (Wang et al., 2019a). In addition, the abundance of microplastics near the shoreline (210 – 240 items/kg dw) may be much higher than in waters farther from the coast (60 – 90 items/kg dw) (Zhang et al., 2019b).

*2.2. Microplastics in freshwater systems*

Microplastics have been widely detected in freshwater systems throughout the world, including Asia, Europe, North America, and Africa (Table 2). The microplastic concentration spans seven orders of magnitude across different samples. In the surface water collected from Tamar Estuary (England), the average abundance of microplastics was 0.028 particles/$m^3$ (Sadri and Thompson, 2014). However, in eight urban lakes in Changsha, China, microplastic concentration in water ranged from 2425 to 7050 particles/$m^3$ (Yin et al., 2019). Similar abundance of microplastics (293 – 7924 particles/$m^3$) was also observed in the surface water of Hong Lake (Wang et al., 2018), Pearl River (Lin et al., 2018), and Nakdong River (South Korea) (Eo et al., 2019). In particular, extremely high concentration of microplastics (172000 to 419000 particles/$m^3$) was detected in Saigon River (Vietnam) (Lahens et al., 2018).

The abundance and distribution of microplastics in rivers and lakes are affected by

human population density, proximity to urban centers, and hydrological and meteorological conditions (Di and Wang, 2018; Kukulka et al., 2012; Thiel et al., 2003). In Wuhan (China), the highest microplastic concentrations were found in the surface waters of Bei Lake (8925 items/$m^3$) and Huanzi Lake (8550 items/$m^3$). Both lakes are located in the very center of the city and are thus surrounded by densely populated residential areas, such that anthropogenic activities likely contribute substantially to the high microplastic concentrations (Wang et al., 2017b). In the Rhine River (Germany), a peak microplastic concentration of 3.9 million particles/$km^2$ was measured in the Rhine-Ruhr metropolitan area, again pointing to the impact of humans (Mani et al., 2015). Similarly, the distribution of microplastics in Pearl River (China) also showed positive correlation with population density and the gross domestic product (Fan et al., 2019). Nevertheless, in Lake Khövsgöl (Mongolia), *i.e.*, a remote area with a low population density, the microplastic concentration (44435 particles /$km^2$) is much higher than in Lake Huron (6541 particles /$km^2$) and Lake Superior (12645 particles /$km^2$), both located in highy developed and densely populated regions between the USA and Canada (Eriksen et al., 2013; Free et al., 2014). The authors proposed that the high microplastic concentration in Lake Khövsgöl might be due to its longer water residence time. In this situation, larger plastics break into microplastics, which continuously accumulate in the lake. The concentration of microplastics in Yangtze River ranged from $1.95 \times 10^5$ to $9.00 \times 10^5$ items/$km^2$, and was also negatively correlated with the velocity of water flow (Xiong et al., 2019). Similarly, in Laurentian Great Lakes, the confluence of currents led to the accumulation of microplastics in the surface water of Lake Erie, which accounts for ~90% of

all the plastic debris detected in all samples (Eriksen et al, 2013). Fischer et al. (2016) reported that the distribution and abundance of microplastics in Lake Chiusi and Lake Bolsena (Italy) were influenced by water circulation. In addition, meteorological conditions also significantly influence microplastic abundance. For example, in the Vuachère and Venoge rivers (both located in Switzerland), the concentrations of microplastics were shown to increase significantly after rain, especially in urbanized areas, whereas in Lakes Zurich and Constance (Switzerland), stronger winds cause vertical mixing and thereby reduce microplastic abundance (Faure et al., 2015).

Freshwater sediments are major sinks of microplastics and they have been widely used as environmental media for the investigation of microplastic pollution (Cozar et al., 2014). A large number of studies showed that microplastics are widely present in freshwater sediments around the world (Table 2). In beach sediment from six Swiss lakes (Geneva, Constance, Maggiore, Neuchatel, Zurich, and Brienz), microplastics with different shapes (foams, fibers, fragments, films, and lines) and chemical composition (PE, PP, and PS) were detected at concentrations ranging from 20 to 7200 particles/$m^2$ (Faure et al., 2015). In a study of the sediments of the Rhine and Main rivers in Germany, the concentration of plastic particles ranged from 228 to 3763 and from 786 to 1368 particles/kg, respectively. PE, PP, and PS particles together accounted for > 75% of all polymer types identified in the sediments, indicative of their extensive use in agriculture, food packaging, and industrial pipelines (Klein et al., 2015). However, as in marine sediments, the abundance of microplastics in freshwater sediments is characterized by spatial variations. A higher mean concentration of

plastic particles in the sediment of Italy's Lake Chiusi was observed at the eastern than at the western shoreline (266 vs. 205 particles/kg dw) (Fischer et al., 2016). The abundance of microplastics in the Rhine sediments in the vicinity of the confluence with the Main River tributary was higher than that before the confluence, which was possibly due to the different flow rate and thus the different mobilization/immobilization of sediments and plastics (Klein et al., 2015). Moreover, the physicochemical characteristics of sediments affect microplastic concentration. Sarkar et al. (2019) reported that the microplastic abundance in the Ganga sediments is highly correlated with the available phosphate content and specific conductivity of the sediments. Overall, microplastics are present in lakes, rivers, and sediments. The role of freshwater systems as transport routes for microplastics to oceans needs to be considered.

*2.3. Microplastics in wastewater and drinking water*

Wastewater treatment plants (WWTPs) are considered as a potential source of microplastics in aquatic environments and the presence of microplastics in the effluents of WWTPs has been demonstrated in many studies (Table 3). In 10 of the largest tertiary WWTPs in Denmark, microplastic concentration in the effluent ranged from 19 to 447 particles/L with a median of 54 particles/L (Simon et al., 2018). Polyester and PE were the most abundant type (Simon et al., 2018). Effluent concentration (in June and October of 2016 and January, April, and July of 2017) across the three secondary WWTPs in South Carolina, USA, ranged from 1 to 30 particles/L (Conley et al., 2019), as comparable to that (28.4 particles/L) observed in a WWTP in Wuhan, China (Liu et al., 2019). In the effluents of a tertiary WWTP in Scotland, UK from May 2017 to February 2018, microplastic

concentration ranged from < 1 to 3 particles/L. Polypropylene was identified as the most abundant polymer type and present as fibers and fragments (Blair et al., 2019). An analysis of 90 effluent samples collected from 17 different tertiary WWTPs across the USA from 2013 to 2015 showed that the concentration of microplastics ranged from 0.004 to 0.195 items/L with an average of 0.05 items/L of treated wastewater. These particles were dominated by fibers and fragments (Mason et al., 2016). Despite the low levels of microplastics on a per liter basis, the estimated daily amount of microplastics released from the effluent was relatively high, with the values at individual facilities ranging from 52773 to nearly $1.5 \times 10^7$ particles per day. Since these facilities process millions of liters of wastewater per day, their effluents are an important source of microplastics in the environment (Mason et al., 2016). Similar amount of microplastics was estimated to be discharged from the effluents of WWTPs in Australia (as high as $1\times10^7$ particles/day) (Ziajahromi et al., 2017), Turkey ($1.2\times10^6$ particles/day) (Gundogdu et al., 2018), and USA ($\sim0.93\times10^6$ particles/day) (Carr et al., 2016).

There have been fewer investigations of microplastics in drinking water. Mintenig et al. (2019) sampled the raw and treated water in drinking water treatment plants and detected microplastics in 10 out of the 24 water samples, with an average concentration in raw water of 0 – 7 particles/$m^3$ and in treated water of 0.7 particles/$m^3$. The size range of the microplastic particles (50 – 150 μm) is conducive to their bioaccumulation, with a possible risk to human health.

## 3. Accumulation of microplastics in aquatic organisms

Ecotoxicology studies have shown microplastic accumulation in a diverse group of

aquatic organisms, including planktonic organisms, invertebrates, and vertebrates (Ribeiro et al., 2019) (Table 4). Nevertheless, evidence for microplastic ingestion in freshwater organisms is much more limited than in marine organisms, both in the number of research conducted and in the number of species investigated. The techniques most commonly used to assess the presence of microplastics in the different tissues and organs of aquatic organisms are optical microscopy, scanning electron microscopy, Raman microscopy, fluorescent microscopy, fluorescent spectroscopy, and Fourier transformed infrared spectroscopy (Ribeiro et al., 2019). However, none provide a rapid, accurate, and quantitative determination of the bioaccumulation kinetics of microplastics. Information on the mechanisms of microplastic bioaccumulation, translocation into organs, cellular transport pathways, and elimination kinetics is also scarce (Wright et al., 2013b).

*3.1. Bioaccumulation*

The accumulation of microplastics in aquatic organisms is determined in part by the physicochemical properties (size and shape) of the particles. In general, smaller microplastics are much more easily ingested. For example, *Acartia clausi* is able to take up considerable amounts of 7.3-µm microplastics but significantly fewer 20.6- and 30.6-µm particles (Cole et al., 2013). The number of planktonic oyster larvae ingesting microplastics and the average amount of microplastics consumed by the larvae decrease with increasing microplastic size (Cole and Galloway, 2015). Oyster larvae of all ages are able to ingest 0.16- to 7.3-µm microplastics but 20.3-µm microplastics are available only to larvae older than 24 days post-fertilization (Cole and Galloway, 2015). Nevertheless, if microplastics further degrade to

nanoscale size, their bioaccumulation is reduced. This was demonstrated in *Daphnia magna*, which was able to take up both 1000- and 20-nm PS microplastics but the uptake rate of the smaller particles was much lower (Rosenkranz et al., 2009). In addition to their size, the shape of microplastics affects their uptake. In their study of the accumulation of microplastics of different shapes (fragments, spheres, and fibers) by the grass shrimp *Palaemonetes pugio*, Gray and Weinstein (2017) found that more fragments were accumulated than spheres and fibers. The uptake of microplastics also varies depending on the life stage of the organism. Brachyuran at zoea are unable to ingest 20.6-μm microplastics, unlike the more developed megalopa (Cole et al., 2013).

*3.2. Distribution*

Microplastics can be taken up by aquatic organisms through several physiological routes and then translocated to different tissues or organs, especially the stomach, intestine, and digestive tract (Magni et al., 2018; Hu et al., 2016). For the clam *Scrobicularia plana*, orally ingested PS microplastics (20 μm) were shown to be transferred to the hemolymph and digestive tract (Ribeiro et al., 2017). The uptake of high-density PE microplastics (< 80 μm) by the mussel *Mytilus edulis* is mediated by ciliary movement on the gill surface, resulting in the transfer of the particles to the digestive system (stomach and intestine) and then to the primary and secondary ducts of the digestive tubules. The particles finally accumulate in the lysosomal system (von Moos et al., 2012). Similarly, Browne et al. (2008) detected PS microspheres (2, 3, and 9.6 μm) in the gut cavity and digestive tubules of *M. edulis* within 12 h of exposure and in the hemolymph and hemocytes 3 days post-exposure. In addition to the

gills, stomach, and intestine, microplastics accumulate in the mussel's gonads, mantle, adductor, viscera, and foot (Kolandhasamy et al., 2018).

Phagocytosis and pinocytosis are two possible pathways for microplastic uptake but particle translocation is largely size-dependent, with smaller plastic particles being more easily internalized. For example, in a study of the zebrafish *Danio rerio*, 5-μm PS microplastics were distributed in the gills, liver, and gut whereas 20-μm PS microplastics accumulated only in the gills and gut (Lu et al., 2016). Jeong et al. (2018) reported the dispersion of 50-nm microplastics in various organs of *Brachionus koreanus* after exposure of this rotifer but 0.5- and 6-μm microplastics localized only to its digestive tract. Surface charge also influences the distribution of microplastics in aquatic organisms. Della Torre et al. (2014) investigated the accumulation of microplastics with different surface coatings (carboxylated and amine polystyrene microplastics) in embryos of the sea urchin *Paracentrotus lividus* and found that while carboxylated PS microplastics were limited to its digestive tract, amine PS microplastics were distributed throughout the embryos.

*3.3. Elimination*

Rapid depuration is a strategy adopted by many aquatic organisms to minimize the potential biological effects of microplastics (Table 5). In the blue mussel *Mytilus galloprovincialis*, 85% and 81% of microplastics were eliminated in low and high microplastic concentration treatments after 144-h depuration, respectively, and 70% of microplastics were eliminated within the first 24 h (Fernández and Albentosa, 2019). Microplastics accumulated in the wild and farmed brown mussel *Perna perna* decreased by

46.8% and 29.0% after 93-h depuration (Birnstiel et al., 2019). *Mytilus edulis* removed 63% of their accumulated microplastics after 6-h depuration (Woods et al., 2018). In the Antarctic krill *Euphausia superba*, the concentration of microplastic beads was shown to decline rapidly, with a depuration rate constant of 0.22 $h^{-1}$. After a 15-day depuration period, only two of the 15 sampled krill contained microplastics (Dawson et al. 2018). A similar depuration pattern was determined in *D. magna* exposed to either pristine or artificial microplastics, as > 90% of the microplastics were eliminated from the daphnid gut after 9-12 min (Ogonowski et al., 2016). The concentration of microplastic fibers in *M. edulis* was likewise shown to decrease significantly after a 48-h elimination phase, although a substantial amount still remained in all of the mussel's organs (Kolandhasamy et al., 2018). The depuration rate depends on the size of the microplastics, with smaller particles often having longer gut retention times than larger ones. In a comparative study, *D. magna* was able to rapidly clear 1000-nm microplastics, such that the particle content decreased by > 90% over 240 min, while the depuration of 20-nm particles over the same time period was only 40% (Rosenkranz et al., 2009). In a study of *Xenopus tropicalis* tadpoles, ~78% of the ingested 10-μm microplastics were eliminated after 1 day, but only 58% of the 1-μm microplastics were depurated at the same time (Hu et al., 2016). Similarly, the depuration of 10-μm and 100-nm PS microplastics from the mussel *M. edulis* and the oyster *Crassostrea virginica* began 6 h and 24 h, respectively, after exposure (Ward et al., 2009). In the Pacific oyster *Magallana gigas*, although the vast majority of ingested microplastic particles were eliminated during the 72-h depuration, 17.7%, 16.7%, and 5.4% of microplastic particles with

the size of 100, 250, and 500 μm, respectively, were still retained in the oysters (Graham et al., 2019). The prolonged residence of smaller particles highlights the need for greater research into the biological effects of microplastics.

**4. Effects of microplastics on aquatic organisms**

Several studies of aquatic organisms have shown the harmful, if not lethal effects of microplastics at the individual, cellular, and molecular levels (Table 4) (Browne et al., 2015; Cole and Galloway, 2015). Effects at the individual level have been demonstrated in *Arenicola marina*, as both its feeding activity and energy reserves were inhibited after microplastic exposure, leading to a decrease in the lugworm's survival (Wright et al., 2013a; Besseling et al., 2013). Decreases in the ingestion rate, feeding capacity, assimilation efficiency, energy budget, and swimming speed of the copepod *Centropages typicus* (Cole et al., 2013; Cole et al., 2015), the marine jacopever *Sebastes schlegelii* (Yin et al., 2018), the oyster *Pinctada margaritifera* (Gardon et al., 2018; Sussarellu et al., 2016), the crab *Carcinus maenas* (Watts et al., 2015), and the diving beetle *Cybister japonicas* (Kim et al., 2018) have been reported as well. Microplastic exposure reduces byssus production, food clearance, and respiration by the Asian green mussel *Perna viridis* and causes abnormal swimming in the goby *Pomatoschistus microps* (Rist et al., 2016; Oliveira et al., 2013). For both organisms, the observed effects can lead to starvation and death. Adverse effects of microplastics on the growth, development, and reproduction of aquatic organisms have also been determined. Growth inhibition and a decrease in the intracellular level of chlorophyll-a were observed in the green alga *Scenedesmus obliquus* exposed to PS microplastics (Besseling et al., 2014). In

*D. magna*, PS microplastics were shown to reduce not only its body size but also the cumulative number of neonates in the first three broods. Also noted in that study were a decrease in the body size of the neonates and various malformations, including incompletely developed antenna setae, a curved shell spine, vacuoles around the ovary, and short antenna setae (Rehse et al., 2016). Gametogenesis, gamete quality, and the fertilization success of oysters are significantly inhibited by microplastics, resulting in decreases in both the total number of oocytes and the velocity of sperm formation as well as mantle or shell malformations (Sussarellu et al., 2016; Tallec et al., 2018).

At the cellular level, microplastic exposure induces multiple adverse effects and stress responses. Embryos of the sea urchin *P. lividus* exposed to 50-nm amino-modified PS microplastics developed a thickened ectodermal membrane and showed abnormal proliferation at 6 h post-fertilization (hpf), followed by striking larval malformation at 48 hpf, including incomplete or absent skeletal rods, a fractured ectoderm, and a reduced arm length (Della Torre et al., 2014). In the brine shrimp *Artemia parthenogenetica* and the zebrafish *D. rerio* exposed to microplastics, abnormalities in intestinal epithelial cell ultrastructure were observed, including a decreased number of microvilli, an increased number of mitochondria, and the splitting of villi and of enterocytes, respectively (Wang et al., 2019c; Lei et al., 2018). In addition to developmental defects, microplastics have been shown to induce oxidative damage, inflammatory responses, and neurotoxicity in rotifers, crabs, mussels, fish, and coral (Jeong et al., 2018; Yu et al., 2018; Brandts et al, 2018a; Ding et al., 2018; Tang et al., 2018). Microplastics also repress detoxification and the immune system of the scleractinian coral

*Pocillopora damicornis* (Tang et al., 2018). Changes in the metabolomic profiles and microbiome diversity of larval zebrafish exposed to PS microplastics have been demonstrated as well (Wan et al., 2018).

At the molecular level, microplastics alter the expression of genes associated with a diverse set of functions. For example, microplastics up-regulate the expression of genes related to the stress response in gilthead seabream *Sparus aurata* L., the nematode *C. elegans* and juveniles of the Chinese mitten crab *Eriocheir sinensis* (Espinosa et al., 2017; Lei et al., 2018; Yu et al., 2018). In juvenile *E. sinensis*, microplastic exposure induces the MAPK signaling pathway via ROS formation, which suggests the enhanced expression of oxidative stress genes (Yu et al., 2018). The expression of genes related to zymogen granules, sterol transport, and the JNK and EGF-ERK1/2 signaling pathway may be up- or down-regulated in the coral *P. damicornis* exposed to microplastics, indicating that, at least in this species, the particles can function as an environmental hormone that modulates physiological activities (Tang et al., 2018). The exposure of *Mytilus galloprovincialis* and *Dicentrarchus labrax* to microplastics was shown to alter the expression of genes related to biotransformation, DNA repair, the stress response, immunity, and lipid metabolism signaling pathways (Brandts et al., 2018a; Brandts et al., 2018b). Changes in gene expression in larvae of the zebrafish *D. rerio* exposed to microplastics are also associated with microplastic exposure (LeMoine et al., 2018).

The toxicity of microplastics may depend on the size, shape, and surface coating of the particles. In the study of Jeong et al. (2016), 0.05-μm PS microbeads had the most deleterious

effects on the growth, fecundity, and lifespan of the rotifer *B. koreanus*, followed by PS microbeads of 0.5 and 6 μm. Snell et al. (2011) and Sjollema et al. (2016) similarly showed that the ability of microplastics to inhibit the growth of microalgae and the reproductive rate of rotifer increases as the particle size decreases. In terms of the shape and surface charge of microplastics, in the grass shrimp *P. pugio* the toxicity of fibers is higher than that of fragments and spheres (Gray and Weinstein, 2017). In a comparison of carboxylated and amine PS microplastics, the former had no effect on the development of sea urchin embryos (2.5-50 μg/mL) whereas the latter caused severe developmental defects and apoptosis (Della Torre et al., 2014).

## 5. Combined effects of microplastics with other pollutants

The large specific surface area of microplastics facilitates their adsorption of organic pollutants, as demonstrated for polybrominated diphenyl ethers (PBDEs), polycyclic aromatic hydrocarbons (PAHs), polychlorinated biphenyls (PCBs), endocrine disruptors, and heavy metals (Chua et al., 2014; Endo et al., 2005; Rochman et al., 2014; Holmes et al., 2012). This ability of microplastics to serve as vectors can result in the bioaccumulation of these and potentially other pollutants. Moreover, plastics themselves contain various additives (e.g., bisphenol A, brominated flame retardants, phthalates, triclosan, bisphenone, and organotins) that can leach into the aquatic environment to cause endocrine-disruptive, carcinogenic or mutagenic effects in aquatic organisms.

### *5.1. Microplastics and organic pollutants*

The few studies on the interaction of microplastics and organic pollutants have

demonstrated multiple effects on the bioaccumulation and toxicity of a wide range of compounds. On the one hand, microplastics can act as carriers of organic pollutants, thus enhancing their bioaccumulation and toxicity. In red tilapia, mussels, *D. magna*, and zebrafish, the accumulation of roxithromycin, pyrene, phenanthrene, and bisphenol A was enhanced in treatments with compared to without microplastics (Zhang et al., 2019c; Avio et al., 2015; Ma et al., 2016; Chen et al., 2017). On the other hand, microplastics can act as a sink for pollutants and thereby reduce their potential bioaccumulation. This was the case in the marine amphipod *Allorchestes compressa*, in which microplastics inhibited the uptake of PBDEs as a result of the adsorption of the pollutant to the particles (Chua et al., 2014). Besides the dual impacts of microplastics on the uptake of organic pollutants, they were also shown to promote the elimination of 2, 2′, 4, 4′-tetrabromodiphenyl ether (BDE-47) by the sandhopper *Talitrus saltator* (Scopetani et al., 2018), which has additional effects on the final bioaccumulation of organic pollutants.

The interactions between microplastics and organic pollutants may be synergistic, additive, or antagonistic. The exposure of the bivalve *Corbicula fluminea* to a mixture of florfenicol and microplastics resulted in synergistic toxicity compared to exposure to the chemical alone (Guilhermino et al., 2018) whereas additive effects were demonstrated for the combination of 50-nm PS microplastics and phenanthrene in *D. magna* (Ma et al., 2016). In *Microcystis aeruginosa*, amino-modified PS microplastics were shown to reduce the toxicity of glyphosate, consistent with the occurrence of antagonistic effects (Zhang et al., 2018). Microplastics also alleviated the neurotoxicity and oxidative damage caused by

roxithromycin in the freshwater fish red tilapia *O. niloticus* (Zhang et al., 2019c). Moreover, the physicochemical properties (e.g., size) of microplastics play an important role in their effects on bioaccumulation and toxicity of organic pollutants. The median effect concentration of phenanthrene didn't change in the presence of 10 μm microplastics, while 50 nm microplastics enhanced the toxicity of phenanthrene to *D. magna* (Ma et al., 2016). However, due to the lack of quantitative methods, the bioaccumulation dynamics of microplastics and organic pollutants are far from clear and the mechanisms underlying the different impacts on toxicity are still unknown.

*5.2. Microplastics and inorganic pollutants*

Most studies on the combined effects of microplastics and inorganic pollutants have focused on heavy metals. Both PS and PVC accumulate metals in seawater, with metal concentrations on plastics up to 800 times higher than those in the surrounding seawater (Brennecke et al., 2016). By concentrating heavy metals, microplastics may alter their bioaccumulation and toxicity. A study of silver uptake by the zebrafish *D. rerio* found a remarkable decrease in the treatment with microplastics compared to the control treatment without microplastics (Khan et al., 2015). In addition, microplastics altered the localization of the metal, causing more of the silver to concentrate in the intestine than in other body tissues of the zebrafish (Khan et al., 2015). While the presence of microplastics was shown to inhibit the accumulation of Cd in the discus fish *Symphysodon aequifasciatus*, treatments combining the two pollutants had antagonistic effects on catalase and phosphatase activities (Wen et al., 2018a). Other studies have shown that microplastics enhance the bioaccumulation and

toxicity of heavy metals. The accumulation of Cd in the liver, gut, and gills of zebrafish increased after co-exposure to microplastics, with greater effects on glutathione, superoxide dismutase, and metallothionein compared to Cd exposure alone (Lu et al., 2018). Similarly, Barboza et al. (2018) found that microplastics enhance the concentration of mercury in the gills and liver of *D. labrax* juveniles by exerting either additive or synergistic effects. The combined effects of microplastics and heavy metals also depend on the physicochemical properties of the plastics. Kim et al. (2017) reported that bare PS microplastics reduce the toxicity of Ni while an opposite trend was found for carboxyl-modified PS particles. Both 0.1-μm and 20-μm PS microplastics enhanced Cu accumulation in zebrafish, while Cu accumulation in the livers and guts in the presence of 0.1-μm PS microplastics was 1.5 – 2.1 times higher than that in the treatment with 20-μm microplastics (Qiao et al., 2019).

## 6. Conclusions and future perspectives

This review provides a summary of current knowledge on the occurrence, accumulation, and toxic effects of microplastics in aquatic organisms. As reviewed, microplastics occur ubiquitously in aquatic environments and a lot of aquatic organisms are exposed to microplastics, which may cause various biological effects. Current research on the bioaccumulation and effects of microplastics mainly focus on marine biota. Information on freshwater organisms is much less well known. Many studies have shown that the accumulation and harmful effects of microplastics in biota depend on the physicochemical properties of the particles (e.g., size, shape, and surface coating) and on the developmental stage of the organism, but in either case the underlying mechanisms are far from clear.

Given the current situation of microplastic pollution, there are some issues that need to be considered in order to better control the problem of microplastics in aquatic systems. Measures including source control, remediation, and cleanup should be taken. First, more countries should be encouraged not to use microbeads in personal care products and biodegradable plastics such as polylactatide and polyhydroxyalkanoates should be used instead. Second, utilization efficiency and circularity of plastics should be improved. Third, existing wastewater treatment plant should be upgraded to improve the removal efficiency of microplastics and to prevent microplastics from entering aquatic system. Moreover, strategies of cleanup and bioremediation technologies should be developed.

Based on the knowledge gaps identified in this review, further research is needed to obtain accurate determination of the actual occurrence of nano-sized plastics in the environment, which in turn will require the development of quantitative methods of detection. More studies on microplastic effects at environmentally relevant concentrations should be performed. In addition, more systematic investigations of the accumulation and toxicity of microplastics with different physicochemical properties (size, shape, and surface coating) in aquatic organisms of different age, sex, developmental stage, and trophic level are needed. The mechanisms by which the physicochemical properties of microplastics govern their bioaccumulation and toxicity need further investigation. Field and laboratory studies with freshwater organisms were scarce and should be performed widely. Moreover, research about the interactions between microplastics and other pollutants is still in its infancy and further studies will have to assess a broader range of chemical pollutants, microplastics, and aquatic

organisms in order to determine the environmental implications of microplastics and their potential impact on human health. Last but not the least, the ecotoxicity of microplastics should be assessed in more environmentally relevant conditions, such as multiple species exposure.

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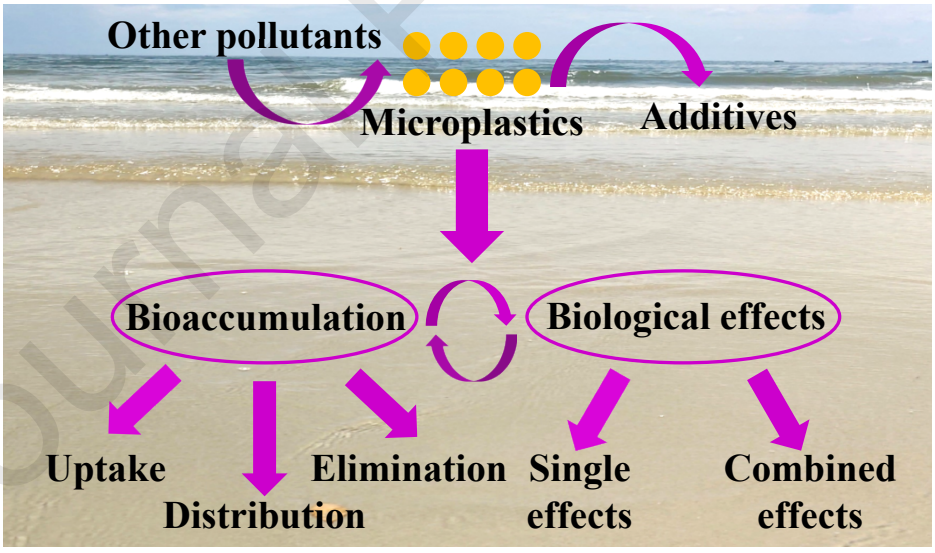
**Figure Legends**

**Figure 1.** Source, transport, distribution, fate and transformation of microplastics in aquatic environment. The black arrows represent the source of microplastics. The blue arrows indicate transport, distribution, fate or transformation of microplastics which can be affected by human activity, hydrological and meteorological conditions.

**Figure 2.** Bioaccumulation and biological effects of microplastics in aquatic organisms as affected by various physicochemical properties of microplastics.

## Graphical Abstract




## HIGHLIGHTS

- Aquatic environments have witnessed high concentrations of microplastics.
- Various aquatic organisms could accumulate microplastics.

- Microplastics may adversely affect aquatic organisms of different trophic levels.
- Combined effects of microplastics and other pollutants have been reported.

**Figure 1**

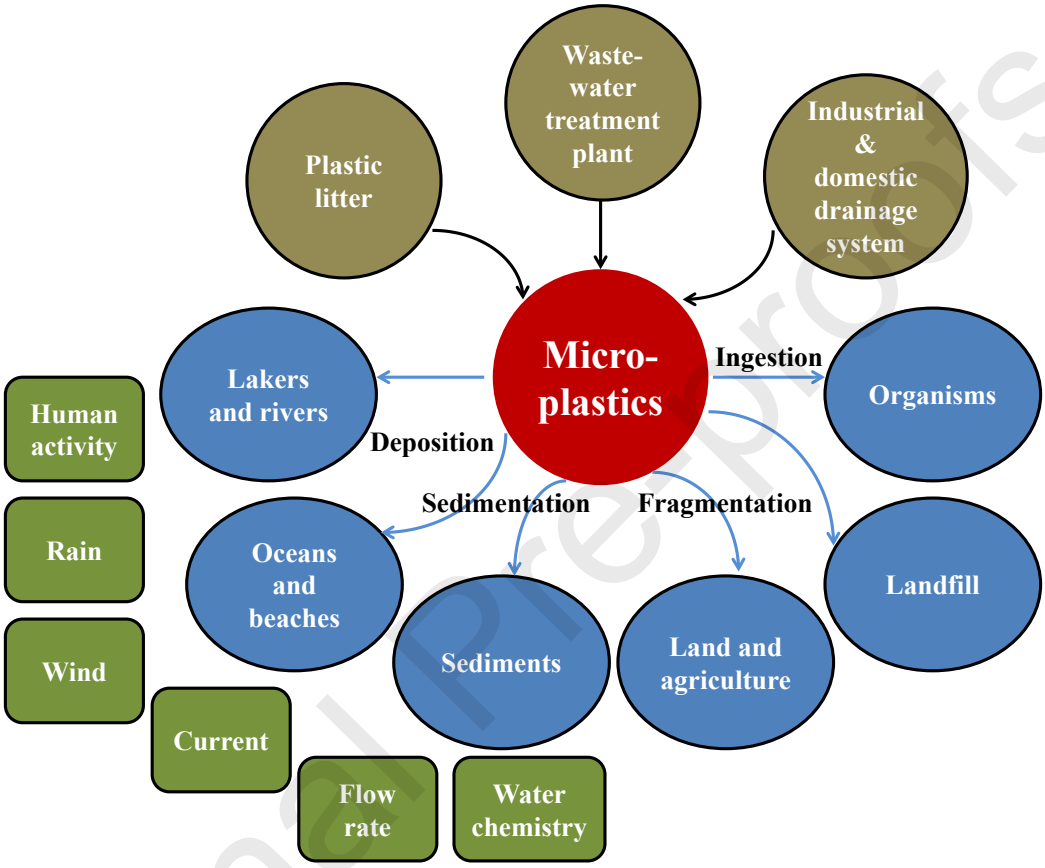


**Figure 2**

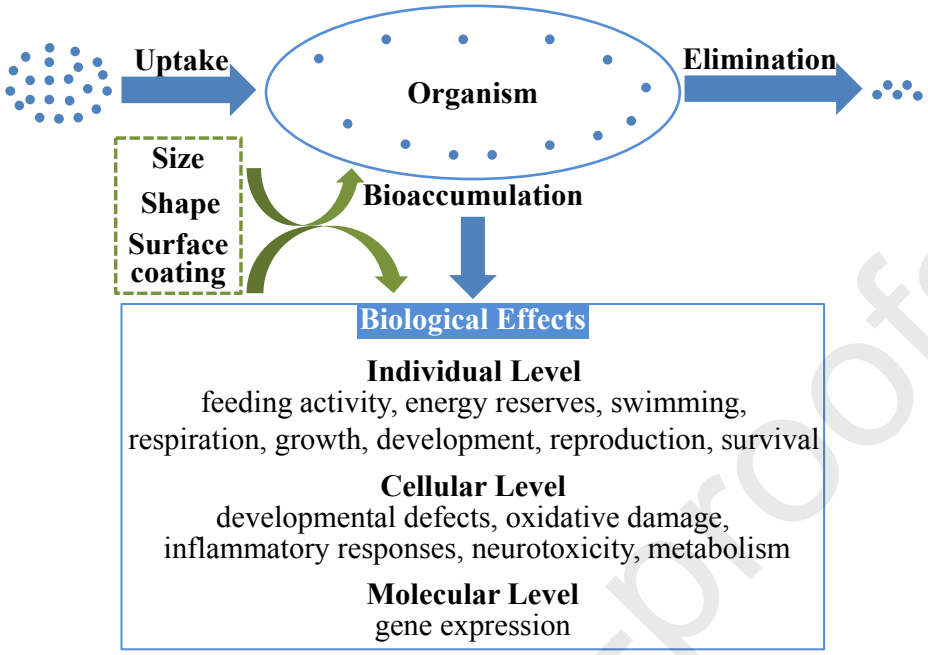



**Table 1.** Distribution of microplastics in seawater and marine sediments around the world (recent studies since 2012).

| Location | Marine environment | Concentration | Unit | Reference |
|---|---|---|---|---|
| Northwestern Pacific Ocean | surface seawater | 640-42000 | items/km$^2$ | (Pan et al., 2019) |
| Arabian Bay | surface seawater | 4.38×10$^4$-1.46×10$^6$ | particles/km$^2$ | (Abayomi et al., 2017) |
| Mediterranean | surface seawater | 0-9.0 × 10$^5$ | items/km$^2$ | (Collignon et al., 2012) |
| Coast of South Korea | surface seawater | 1051 | particles/m$^3$ | (Song et al., 2018) |
| South and southwest | surface Arctic | 0-1.31 | items/m$^3$ | (Lusher et al., |

| | | | | |
|---|---|---|---|---|
| of Svalbard, Norway | water | | | 2015) |
| South and southwest of Svalbard, Norway | sub-surface Arctic water | 0-11.5 | items/$m^3$ | (Lusher et al., 2015) |
| Northeast Greenland | sub-water | 1-3 | items/$m^3$ | (Morgana et al., 2018) |
| Antarctic Peninsula | seawater | 755-3524 | items/$km^2$ | (Lacerda et al., 2019) |
| Southern North Sea | surface water | 0.1-245.4 | particles/$m^3$ | (Lorenz et al., 2019) |
| Northeast Atlantic Ocean | water | 2.46 | particles/$m^3$ | (Lusher et al., 2014) |
| Coastline of Bandar Abbas | water | 3252 | particles/$m^2$ | (Nabizadeh et al., 2019) |
| Jiaozhou Bay, China | seawater | 20-120 | items/$m^3$ | (Zheng et al., 2019) |
| Bohai Bay, China | surface water | 650.0-2700.0 and 540.0-1550.0 | items/$m^3$ | (Wu et al., 2019) |
| Banderas Bay, Mexico | coastal water | 0.013-0.044 | pieces/$m^3$ | (Pelamatti et al., 2019) |
| Chabahar Bay, Gulf of Oman (Makran Coasts) | surface water | 0.07-1.14 | particles/$m^3$ | (Aliabad et al., 2019) |
| Kingston Harbour, Jamaica | surface water | 0-5.73 | particles/$m^3$ | (Rose and Webber, 2019) |
| Ross Sea (Antarctica) | surface water | 0.0032-1.18 | particles/$m^3$ | (Cincinelli et al., 2017) |
| Northern Gulf of Mexico | surface water | 4.8-18.4 | particles/$m^3$ | (Di Mauro et al., 2017) |

| | | | | |
|---|---|---|---|---|
| North Atlantic Ocean, Scotland | deep-sea water | 70.8 | particles/$m^3$ | (Courtene-Jones et al., 2017) |
| Jinhae Bay, South Korea | surface water | 88 | particles/L | (Song et al., 2015) |
| South-eastern coastline of South Africa | water | 257.9-1215 | particles/$m^3$ | (Nel and Froneman, 2015) |
| Slovenian part of the Northern Adriatic | surface water | $406\times10^3$ | particles/$km^2$ | (Gajšt et al., 2016) |
| Israeli Mediterranean coastal waters | surface water | 7.68 | particles/$m^3$ | (van der Hal et al., 2017) |
| Bohai Sea, China | water | 0.33 | particles/$m^3$ | (Zhang et al., 2017) |
| Maowei Sea, China | sediment | 520-940 | items/kg dw | (Li et al., 2019) |
| Northern Bering and Chukchi Seas | sediment | < 68.88 | items/kg dw | (Mu et al., 2018). |
| South Yellow Sea | sediment | 1765 | items/kg dw | (Wang et al., 2019a) |
| Yellow Sea and East China Sea | sediment | 60-240 | items/kg dw | (Zhang et al., 2019b) |
| Southern North Sea | sediment | 2.8-1188.8 | particles/kg dw | (Lorenz et al., 2019) |
| Beaches in Mumbai, India | sediment | 12-960 | items/$m^2$ | (Jayasiri et al., 2013) |
| Beaches in Slovenia | sediment | 133.3-155.6 | particles/kg dw | (Laglbauer et al., 2014) |
| Norderney, North Sea coast of | sediment | 2.3 | particles/kg dw | (Dekiff et al., 2014) |

| | | | | |
|---|---|---|---|---|
| Germany | | | | |
| Halifax Harbor, Nova Scotia | sediment | 2000-8000 | particles/kg dw | (Mathalon and Hill, 2014) |
| Bohai Bay, China | sediment | 96.7-333.3 and 56.7-113.3 | items/kg dw | (Wu et al., 2019) |
| Beaches of Tenerife (Canary Islands, Spain) | sediment | 2-115.5 | items/$m^2$ | (Alvarez-Hernandez et al., 2019) |
| Beach in the Biosphere Reserve of Lanzarote | sediment | 36.3 | g/$m^2$ | (Edo et al., 2019) |
| Sishili Bay, North Yellow Sea, China | sediment | 499.76 | items/kg dw | (Zhang et al., 2019a) |
| Northwestern Mediterranean Sea | sediment | 33-798 and 12-187 | items/kg dw | (Constant et al., 2019) |
| Beaches in eastern waters of Hong Kong | sediment | 0.58-2116 | items/kg dw | (Lo et al., 2018) |
| South-eastern coastline of South Africa | sediment | 688.9-3308 | particles/$m^2$ | (Nel and Froneman, 2015) |
| Huatulco Bay, Pacific coast of southern Mexico | sediment | 1600-2300 | items/kg dw | (Retama et al., 2016) |
| Northern Gulf of Mexico estuaries | sediment | 2-117 | particles/$m^2$ | (Wessel et al., 2016) |
| Mediterranean Sea, Spain | sediment | 100.78-897.35 | items/kg dw | (Alomar et al., 2016) |

| | | | | |
|---|---|---|---|---|
| Beaches of Kaliningrad region, Russia | sediment | 1.3-36.3 | items/kg dw | (Esiukova, 2017) |
| Coasts of Tyrrhenian Sea, Italy | sediment | 42-1069 | items/kg dw | (Cannas et al., 2017) |
| Baltic Sea, Russia | sediment | 34 | items/kg dw | (Zobkov and Esiukova, 2017) |
| Terra Nova Bay (Ross Sea, Antarctica) | sediment | 1-90 | items/$m^2$ | (Munari et al., 2017) |
| Strait of Hormuz, Persian Gulf | sediment | 2-1258 | particles/kg dw | (Naji et al., 2017a) |
| Persian Gulf, Iran | sediment | 61 | particles/kg dw | (Naji et al., 2017b) |

**Table 2.** Microplastics in freshwater and sediments around the world (recent studies since 2013).

| Location | Freshwater environment | Concentration | Unit | Reference |
|---|---|---|---|---|
| Bei Lake (Wuhan, China) | surface water | 8925 | items/$m^3$ | (Wang et al., 2017b) |
| Huanzi Lake (Wuhan, China) | surface water | 8550 | items/$m^3$ | (Wang et al., 2017b) |
| Rhine River (central Europe) | surface water | 3.9 million | particles/$km^2$ | (Mani et al., 2015) |
| Lake Khövsgöl (Mongolia) | surface water | 44 435 | particles/$km^2$ | (Free et al., 2014) |
| Tamar Estuary (England) | surface water | 0.028 | particles/$m^3$ | (Sadri and Thompson, 2014) |
| Pearl River Estuary (Hong Kong, China) | surface water | 94-2098 | items/$m^2$ | (Fok and Cheung, 2015) |
| Lakes Huron (North America) | surface water | 6541 | particles/$km^2$ | (Eriksen et al., 2013) |
| Lake Superior (North America) | surface water | 12645 | particles/$km^2$ | (Eriksen et al., 2013) |
| Xianjia Lake (China) | surface water | 3825 | items/$m^3$ | (Yin et al., 2019) |
| Meixi Lake (China) | surface water | 2563 | items/$m^3$ | (Yin et al., 2019) |
| Yang Lake (China) | surface water | 2425 | items/$m^3$ | (Yin et al., 2019) |

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|---|---|---|---|---|
| Yue Lake (China) | surface water | 3300 | items/$m^3$ | (Yin et al., 2019) |
| Yuejin Lake (China) | surface water | 7050 | items/$m^3$ | (Yin et al., 2019) |
| Nianjia Lake (China) | surface water | 5600 | items/$m^3$ | (Yin et al., 2019) |
| Dong Lake (China) | surface water | 4113 | items/$m^3$ | (Yin et al., 2019) |
| Donggua Lake (China) | surface water | 5063 | items/$m^3$ | (Yin et al., 2019) |
| Dongting Lake (China) | surface water | 900-2800 | items/$m^3$ | (Wang et al., 2018) |
| Hong Lake (China) | surface water | 1250-4650 | items/$m^3$ | (Wang et al., 2018) |
| Tibet plateau lake (China) | surface water | 8-563 | items/$m^2$ | (Zhang et al., 2016) |
| Hudson River (USA) | surface water | 0.625-2.45 | fibers/L | (Miller et al., 2017) |
| Subalpine Lakes (Italian) | surface water | 4000-57000 | particles/$km^2$ | (Sighicelli et al., 2018) |
| Three Gorges Reservoir (China) | surface water | 1597-12611 | n/$m^3$ | (Di and Wang, 2018) |
| Nakdong River (South Korea) | surface water | 293-4760 | particles/$m^3$ | (Eo et al., 2019) |
| Pearl River (China) | surface water | 379-7924 | items/$m^3$ | (Lin et al., 2018) |
| Saigon River (Vietnam) | surface water | 172000-419000 | items/$m^3$ | (Lahens et al., 2018) |

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|---|---|---|---|---|
| Carpathian basin (Europe) | surface water | 3.52-32.05 | particles/$m^3$ | (Bordos et al., 2019) |
| The Teltow Canal (Germany) | water | 0.01-95.8 | items/L | (Schmidt et al., 2018) |
| Three Gorges Reservoir (China) | sediment | 25-300 | items/kg ww | (Di and Wang, 2018) |
| Nakdong River (South Korea) | sediment | 1971 | particles/kg dw | (Eo et al., 2019) |
| Pearl River (China) | sediment | 80-9597 | items/kg dw | (Lin et al., 2018) |
| Yangtze River Delta (China) | surface water | 0.48-21.52 | items/L | (Hu et al., 2018) |
| Yangtze River Delta (China) | sediment | 35.76-3185.33 | items/kg dw | (Hu et al., 2018) |
| Antuã River (Portugal) | water | 58-193 (March), 71-1265 (October) | items/$m^3$ | (Rodrigues et al., 2018) |
| Antuã River (Portugal) | sediment | 100-629 (March), 18-514 (October) | items/kg dw | (Rodrigues et al., 2018) |
| Carpathian basin (Europe) | sediment | 0.46-1.62 | particles/kg dw | (Bordos et al., 2019) |
| stormwater pond (Denmark) | pond water | $2.7\times10^5$ | items/$m^3$ | (Olesen et al., 2019) |
| stormwater pond (Denmark) | sediment | $9.5\times10^5$ | items/kg dw | (Olesen et al., 2019) |

| | | | | |
|---|---|---|---|---|
| Rhine River (Germany) | sediment | 228-3763 | particles/kg dw | (Klein et al., 2015) |
| Lagoon-Channel of Bizerte (Northern Tunisia) | sediment | 3000-18000 | items/kg dw | (Abidli et al., 2017) |
| Bizerte (Northern Tunisia) | sediment | 2340-6920 | items/kg dw | (Toumi et al., 2019) |
| Main River (Germany) | sediment | 786-1368 | particles/kg dw | (Klein et al., 2015) |
| River Tame (UK) | sediment | 165 | particles/kg dw | (Tibbetts et al., 2018) |
| Lakes and rivers in Changsha (China) | sediment | 270.17-866.59 | items/kg dw | (Wen et al., 2018b) |
| Taihu Lake (China) | surface water | 3.4-25.8 | items/L | (Su et al., 2016) |
| Taihu Lake (China) | sediment | 11.0-234.6 | items/kg dw | (Su et al., 2016) |
| Beijiang River | sediment | 178-544 | items/kg dw | (Wang et al., 2017a) |
| Vembanad Lake (India) | sediment | 252.80 | particles/$m^2$ | (Sruthy and Ramasamy, 2017) |
| Japan, Thailand, Malaysia, and South Africa | sediment | 100-1900 | pieces/kg dw | (Matsuguma et al., 2017) |
| South Carolina Estuaries (USA) | sediment | 413.8 (Charleston Harbor); 221.0 (Winyah Bay) | particles/$m^2$ | (Gray et al., 2018) |

| | | | | |
|---|---|---|---|---|
| Shanghai, China | sediment | 802 | items /kg dw | (Peng et al., 2018) |

**Table 3.** Summarized researches from 2016-2019 on microplastics in the effluent of WWTPs.

| Location | WWTP type | Microplastic concentration | Most common polymer found | Reference |
|---|---|---|---|---|
| UK | tertiary | <1-3 particles/L | fragment, fibers, film, PP, PE, PVS, POM | (Blair et al., 2019) |
| UK | secondary | 0.25-8.7 particles/L | flakes, fibers, film, beads, and foam | (Murphy et al., 2016) |
| USA | secondary | 1-30 particles/L | fibers and particle | (Conley et al., 2019) |
| USA | tertiary | 0.05 particles/L | fibers and fragment | (Mason et al., 2016) |
| USA | tertiary | $8.8\times 10^{-4}$ particles/L | polyethylene particles | (Carr et al., 2016) |
| China | secondary | 28.4 particles/L | fiber and fragment | (Liu et al., 2019) |
| China | secondary | 0.13 and 0.05 particles/L, | fragment, fibers, film | (Lv et al., 2019) |
| South Korea | tertiary | 33-297 particles/L | microbeads and fragments | (Hidayaturrahman and Lee, 2019) |

| | | | | |
|---|---|---|---|---|
| Australia | primary | 1.54 particles/L | PET fibers and irregular shaped PE particles | (Ziajahromi et al., 2017) |
| Australia | secondary | 0.48 particles/L | PET fibers and irregular shaped PE particles | (Ziajahromi et al. 2017) |
| Australia | tertiary | 0.28 particles/L | PET fibers and irregular shaped PE particles | (Ziajahromi et al. 2017) |
| Netherlands | tertiary | 9-91 particles/L | fibers, spheres and foils | (Leslie et al., 2017) |
| Germany | secondary | 0.08-7.52 particles/L | PE, PP | (Mintenig et al., 2017) |
| Denmark | tertiary | 19 particles/L | PE, PS | (Simon et al., 2018) |
| Canada | secondary | 0.5 particles/L | fibers, fragments | (Gies et al., 2018) |
| Turkey | secondary | 1249102 particles/day | fibers, film, fragment | (Gundogdu et al., 2018) |
| Turkey | secondary | 351019 particles/day | fibers, film, fragment | (Gundogdu et al., 2018) |
| Finland | tertiary | 0.02-0.3 particles/L | polyester, PE, polyacrylate | (Talvitie et al., 2017) |

| | | | | |
|---|---|---|---|---|
| Finland | secondary | 0.4-1 particles/L | polyester, PE | (Lares et al., 2018) |

**Abbreviations:** PP=polypropylene, PE=polyethylene, PVS=polyvinyl stearate, POM=polyoxymethylene, PET= polyethylene terephthalate, PS=polystyrene.

**Table 4.** Accumulation and ecotoxicological effects of microplastics in aquatic organisms in laboratory.

| ganism | Microplastic type | Microplastic size | Microplastic concentration | Ecotoxicological effects | Reference |
|---|---|---|---|---|---|
| on taxa | PS | 0.4-30.6 μm | 3000 beads/mL (7.3 μm); 2240 beads/mL (20.6 μm); 635 beads/mL (30.6 μm) | ingestion rates decreased | (Cole et al., 2 |
| ster | fluorescent PS, aminated PS ($PS-NH_2$), carboxylated PS (PS-COOH) | 0.07, 0.16, 0.87, 1.84, 4.1, 7.3, 10.2, and 20.3 μm fluorescent PS 0.99 μm $PS-NH_2$, 0.94 μm PS-COOH | 1, 10, 100, or 1000 particles/mL | ingestion rates decreased (1 μm PS) | (Cole et al., 2 |
| np *etes* | PS, PP, PE | 30,75 μm PS 34, 93 μm PP 35, 59, 83, 116, 165 μm PE | 50 000 particles/L | 20%-55% mortality | (Gray and We 2017) |
| *ria* | PS | 20 μm | 4 particles/mL | antioxidant capacity changed, DNA damage, neurotoxicity and oxidative damage | (Ribeiro et al |
| el *ulis* L. | HDPE | <80 μm | 2.5 g/L | granulocytomas formation increased, lysosomal membrane stability decreased | (von Moos et |
| L. | PS | 2, 3 and 9.6 μm | 0.51 g/L | no significant biological effects | (Browne et al |
| el | microfibers | / | 2000 microfibers/L | / | (Kolandhasar 2018) |
| *a* | PS | 10 μm and 1 μm | 50 mg/L | significant modulation of CAT and GPx activities, significant increase of DOP level | (Magni et al., |

| | | | | | |
|---|---|---|---|---|---|
| | PS | 10 μm and 1 μm | 10, $10^3$ and $10^5$ particles/mL (1 μm);<br>0.1, 10 and $10^3$ particles/mL (10 μm), | / | (Hu et al., 20 |
| *o* | PS | 5 and 20 μm | 20 mg/L | inflammation, oxidative stress, alterations of metabolic profiles in liver, disturbed the lipid and energy metabolism | (Lu et al., 201 |
| *s* | PS | 0.05, 0.5 and 6 μm | 10 mg/L | oxidative stress and lipid peroxidation | (Jeong et al., |
| *otus* | PS | 40 nm PS-COOH<br>50 nm PS-NH2 | 2.5, 5, 10, 25 and 50 μg/mL PS-COOH ;<br>1, 2.5, 3, 5, 10 and 50 μg/mL $PS-NH_2$ | malformation of embryos, modulation of gene expression | (Della Torre e 2014) |
| krill | PE | 27-32 μm | 29 beads/mL or 400 ng/mL | no mortality and weight loss | (Dawson et a |
| *nagna* | PE | 1-5 μm | $10^4$ particles/mL | delevated mortality, increased inter-brood period, decreased reproduction and food intake | (Ogonowski e 2016). |

**Abbreviations:** PS=polystyrene, PP=polypropylene, PE=polyethylene, HDPE=high-density polyethylene.

**Table 5**. Elimination of microplastics by aquatic organisms in laboratory.

| Species | Depuration period | Depuration conditions | Percentage of elimination | Reference |
|---|---|---|---|---|
| brown mussels (*Perna perna*) | 93 h | seawater filtered by 11 μm filter paper, no water exchange, no feeding | 46.79% (wild), 28.95% (farmed) | (Birnstiel et al., 2019) |
| *Mytilus galloprovincialis* | 144 h | seawater filtered by 0.45 μm filter, water exchange, feeding | 85% and 81% | (Fernández and Albentosa, 2019) |
| *Mytilus edulis* | 2 h | seawater filtered through 1μm filter | very little or no egestion | (Rist et al., 2018) |
| *Mytilus edulis* | 6 h | 0.2 μm filtered seawater changed daily, no feeding | 63% | (Woods et al., 2018) |
| *Mytilus edulis* | 48 h | water filtered through 0.45 μm filter paper | / | (Kolandhasamy et al., 2018) |
| *Euphausia superba* | 20 h | filtered by 0.2 μm filter, no water changes, no feeding | 77.8% | (Dawson et al., 2018) |
| *Oryzias melastigma* | 7 d | feeding | 86.9% and 89.6% | (Cong et al., 2019) |
| *Cybister japonicus* | 48 h | feeding | 100% | (Kim et al., 2018) |
| *Artemia parthenogenetica* | 14 d | artificial seawater filtered by 0.22 μm filter, feeding, water exchange | 94% | (Wang et al., 2019b) |

| | | | | |
|---|---|---|---|---|
| *Daphnia magna* | 12 min | M7 medium, no feeding, no water exchange | >90% | (Ogonowski et al., 2016) |
| *Daphnia magna* | 240 min | medium changed hourly, no food | > 90% (1000 nm), 40% (20 nm) | (Rosenkranz et al., 2009). |
| *Magallana gigas* | 72 h | seawater filtered through GF/F filter, no feeding | 82.3% (100 µm); 83.3% (250 µm); 94.6% (500 µm) | (Graham et al., 2019) |
| *Xenopus tropicalis* | 6 d | no feeding, filtered water | >95% | (Hu et al., 2016) |