Microplastics in Freshwater Environments

Lorena M Rios Mendoza and Mary Balcer, Department of Natural Sciences, University of Wisconsin-Superior, Superior, WI, United States

Introduction 1
Classification and Types of MPs 2
MP Sources 2
MP Collection and Identification 4
Distribution of MPs in Freshwaters 11
MPs and Toxic Compounds 11
Interactions of Aquatic Organisms with MPs 12
MPs as a Habitat for Microbes 12
Field Studies on Ingestion and Other Interactions 13
Laboratory Studies 13
Summary 24
References 24
Further Reading 29

Abstract

The constantly increasing production and use of synthetic plastic materials are responsible for the global increase in plastic debris in freshwater environments. Microplastics (MPs) have been found in the water and sediments of lakes and rivers worldwide, with concentrations correlated with human population density. Wastewater treatment plants have been identified as one of the main point sources for MP release, including microbeads and clothing fibers. Synthetic polymers are identified by their chemical composition using Fourier Transform Infrared and Raman spectroscopies. In freshwaters, the primary synthetic polymers that have been detected are polyethylene, polypropylene, polyester, and polystyrene. Field studies show that MPs are ingested by a variety of freshwater organisms, including clams, insects, and fish, but are retained at relatively low densities. Laboratory experiments document the effects that MPs can have on the metabolism, growth, survival, and reproduction of aquatic organisms, but usually at plastic particle concentrations that are orders of magnitude higher than those currently found in nature. The increased surface area of the small MP particles allows them to concentrate persistent organics pollutants from the aquatic environment. At this time there is limited information available on the concentration of toxic chemicals on freshwater MPs or the kinetics of the release of these compounds to organisms upon ingestion. Because MPs do not readily degrade, it is important for humans to reduce our reliance on plastic materials before their concentration in freshwater biomes reaches critical levels.

Introduction

The extensive use of plastic items worldwide can be attributed to plastics' flexible properties and low costs of production. For example, the physical characteristics of polyethylene and polypropylene give plastic items high resistance to aging and prevent deterioration (Derraik, 2002). Worldwide, plastic production has increased dramatically since the 1950s, reaching the current rate of 359 million metric tons per year (Plastics Europe, 2019). The cumulative production of plastic from 1950 to date has been calculated as over 8300 million metric tons (Blair et al., 2019). This excess production and use of plastic items have resulted in the designation of this era as the *Plasticene Age* or *Age of Plastics* (Reed, 2015). The production of plastic materials is expected to double in the next 20 years. With less than 25% of plastic items being recycled (Alimi et al., 2018), the potential for environmental pollution by plastic debris is intensifying. The fact that most plastics are not biodegradable on any reasonable human time scale increases the threat of environmental impact from plastic. Although plastics can photodegrade (UV-B degradation) and be broken down by mechanical abrasion, these processes end up producing massive amounts of small particles of microplastics (MPs) that continue to persist in the environment. An estimated eight million tons of plastic debris enters the oceans every year (Jambeck et al., 2015), but there is currently little information on the total amount of plastic accumulating in freshwater biomes.

MPs are small particles of synthetic polymers derived from petroleum and are produced intentionally (e.g. preproduction pellets) and unintentionally (e.g. fragmentation of larger plastic debris). MPs are a relatively new type of environmental contaminant whose impacts on ecosystems are very poorly understood. While MPs have received substantial attention in the marine environment, in freshwater biomes these particles have only been documented and studied during the last decade. MP particles have now been reported from freshwaters worldwide and an increasing number of studies are being conducted to explore the adverse effects that MP may have on aquatic organisms and ultimately on human health.

MPs are ubiquitous anthropogenic stressors affecting all of the world's biomes. MPs are found in the air (Panko et al., 2013; Sommer et al., 2018), water, and soil, and can easily be ingested by organisms (Browne et al., 2011). Due to their small sizes, MPs

 https://doi.org/10.1016/B978-0-12-409548-9.12394-2

can concentrate toxic compounds from the environment (Rodrigues et al., 2019). Additionally, some of the chemical additives used during plastic production are endocrine disrupters (Monneret, 2017). There is concern that MPs may also affect humans through ingestion, inhalation, or dermal contact (Prata et al., 2019b).

Classification and Types of MPs

Currently, there is no consensus on the definition of MPs based on size that is congruent with SI (System International) units. To distinguish macro and micro plastic particles, 5 mm was proposed as the maximum size limit for MPs (Arthur et al., 2009; Barnes et al., 2009; Ryan et al., 2009; Thompson et al., 2009). However, some researchers have used different size classifications for macro (1–100 cm), meso (1–25 mm), micro (1–5000 μm) and nanoplastics (1–100 nm or 1–20 μm) (e.g. Hartmann et al., 2019). Frias and Nash (2019) recently defined MPs as "any synthetic solid particle or polymeric matrix, with regular or irregular shape and with size ranging from 1 μm to 5 mm in maximum dimension, of either primary or secondary manufacturing origin, which are insoluble in water." Whereas, the Group of Experts on the Scientific Aspects of Marine Environmental Protection (GESAMP, 2015) has noted the lack of international agreement on MP definitions, they currently recommend the use of MPs for all particles <5 mm in diameter. For this paper, we are only including reports of MP particles ranging from 1 to 5000 μm.

There are more than 5000 types of commercial plastic polymers (Wagner and Lambert, 2018), however, six polymers make up approximately 80% of plastic production: polypropylene (PP), polyethylene (PE), polyvinyl chloride (PVC), polyethylene terephthalate (PET, also known as polyester PES), polystyrene (PS), and polyurethane (PU), Table 1 (Utracki, 2003; Plastics Europe, 2019). Plastics with a density less than 1.0 g mL^{-1} will float in freshwaters, while those with higher densities tend to sink (Andrady, 2011).

MPs are found in a variety of shapes in aquatic biomes including filaments/fibers, films, fragments, spheres/pellets and foams and occur in a range of colors including clear, white, red, brown, yellow, green, blue and black (Rodríguez-Seijo and Pereira, 2017). MP fibers are increasingly reported as one of the most common forms of MPs in samples from freshwater environments (Eerkes-Medrano and Thompson, 2018). Figs. 1 and 2 are showing MPs found in samples form environmental and cosmetic products.

MP Sources

MPs can be produced in two distinct ways. Primary MPs are plastic particles that are initially manufactured in small sizes, including virgin resin pellets or flakes and preproduction plastics. Synthetic microbeads are used in personal care products (facial cleansers,

Table 1 Some main types of plastics and their uses.

Name	*Abbreviation*	*Density (g mL^{-1})*[a]	*Uses*
Thermoplastics			
Polypropylene	PP	0.90	Food packing, snack wrappers, automotive parts, bank notes
High density polyethylene	HDPE	0.96	Toys, milk bottles, shampoo bottles, houseware, pipes
Low density polyethylene	LDPE	0.92	Bags, trays, containers, agricultural films
Polyvinyl chloride	PVC	1.40	Windows frames, floor and wall covering, pipes, cables
Polyethylene terephthalate[b]	PET/PETE/PEST	1.55	Soda and water bottles, clothing
Polystyrene	PS/PST	<1.05	Expanded foam, eyeglasses frames, egg trays, insulation, cups
Polyamides (nylons)	PA	1.02–1.14	Adhesives, fibers, films, clothing
Polycarbonate	PC	1.36	Electrical connectors, insulators, medical tubing, instrument covers
Polymethylmethacrylate	PMMA	1.18	Lightweight sheets or Plexiglass™
Thermosets			
Polyurethane	PU/PUR	1.052	Building insulation, pillows and mattresses, fridge insulations
Epoxy resins	EP	1.82	Adhesives, glass reinforced, paints and coatings
Polyvinyl ester	PVE	1.80	Paints, transportation, buildings, aerospace,
Cellulose acetate	CA	1.30	Containers, tools, packaging, fibers
Silicone		1.5	Seals, O-rings, linings, insulations
Phenol formaldehyde	PF	1.21	Bakelite™, billiard balls, paints, plywood
Urea formaldehyde	UF	1.5	Decorative laminates, textiles, cotton blends, electrical appliances
Phenolic resins	PH	1.28	Laboratory countertops, coating, adhesives, billiard balls
Acrylic resins	AC	1.19	Automotive, architectural, coating

[a]Sea water density 1.02–1.03 g mL^{-1}, freshwater density ~1.0 g mL^{-1}.
[b]Polyethylene terephthalate is also known as polyester (PES).
Data from Utracki, L. A. (2003). *Polymer blends handbook.* USA: Kluwer Academic Publishers; Plastics Europe. (2019). Plastics-the facts. An analysis of European plastics production, demand and waste data. Plastics Europe: Association of Plastic Manufacturers.

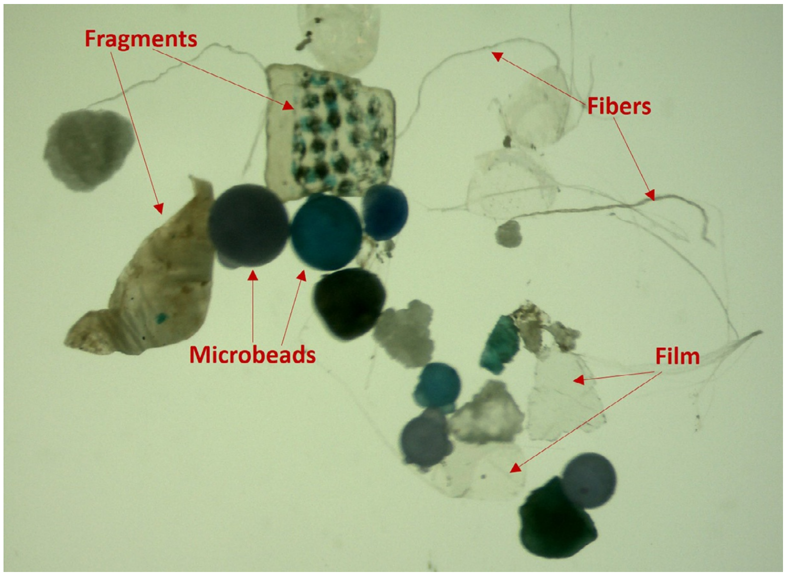

Fig. 1 Sample from St. Louis River Estuary, Duluth, MN showing MP types.

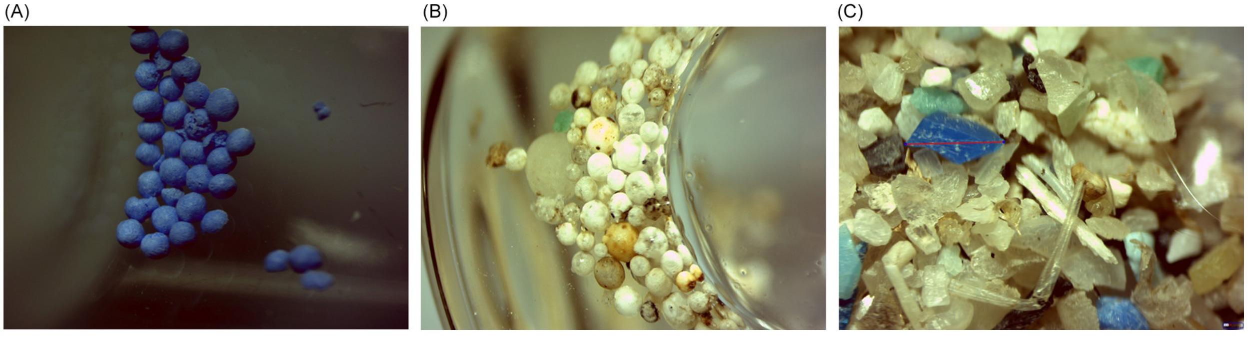

Fig. 2 Microplastics from (A) cosmetic products with microbeads (bar size 670 μm), (B) freshwater environment (pellet size 4988 μm, Lake Superior) and (C) marine environment (fragment size 2149 μm, Mediterranean Sea).

tooth pastes, shower gels, makeup, insect repellents and sunscreens) and as carriers for certain types of medicine. They occur in powders used for injection molding and in ink for 3D printers. Primary MPs are frequently used as industrial abrasives for cleaning using air/water-blasting, and are found in drilling fluids employed in oil and gas exploration (Duis and Coors, 2016).

Secondary MPs are produced by fragmentation of larger plastic items as a result of photodegradation (UV radiation) and physical abrasion which produce small particles that fall into the micro- or even nanoparticle size ranges. Sources of MP fragments include the breakdown of plastic bags, bottles, cups, food wraps, plastic films used to cover agricultural soil and crops, foam insulation materials, and even from automobile tires (Duis and Coors, 2016; Wagner and Lambert, 2018). Synthetic microfibers are created from degradation of clothing and textile products containing blends of polyester, polyacrylic, and nylon (Browne et al., 2011).

Plastic materials can enter aquatic systems from accidental industrial spills, runoff from agricultural fields that use plastic mulches, inadequate recycling, discharges from municipal wastewater treatment plants, and individual littering. Microbeads that are used in cosmetics and toothpastes often end up in domestic wastewater. Kalčíková et al. (2017) reported the daily release of an average of 15.2 mg of microbeads per person to the sewage system in Ljubljana, Slovenia. MP fibers produced from laundering textiles are another major source of MP pollution. Washing a normal 6 kg load of polyester-cotton blend clothing can release 138,000 fibers, while 496,000 fibers can be produced from a load of polyester clothing, and 729,000 fibers from a load of acrylic materials (Napper and Thompson, 2016). The initial washing of fabrics releases the largest amount of fibers, but fibers continue to released even after ten washings (De Falco et al., 2019).

While there are many sources of MPs in freshwater systems, wastewater treatment plants (WWTPs) are one of the most important point sources for microfibers and microbeads (Carr et al., 2016; Dris et al., 2018; Ou and Zang, 2018). WWTPs were developed with the objective of treating human and industrial wastes to remove organic matter, nutrients, and harmful bacteria so that treated effluents could be discharged into marine and freshwater systems without compromising human health or causing environmental damage (Vijayan and Mohan, 2013). The plants are not specifically designed to remove MPs and during treatment can actually form additional MP sized particles when larger particles are abraded as the plastic debris moves through the plant.

Studies have examined the effectiveness of plants with primary, secondary, and tertiary treatment methods in removing MPs (Table 2). Influent waters often contain between 12 and 7000 plastic particles (pp) L^{-1} (Gündoğdu et al., 2018; Simon et al., 2018). Tertiary WWTPs are able to remove up to 98% of the microliter (natural and synthetic) during treatment which includes screening, grit removal, and chemically enhanced settling (Talvitie et al., 2017). Smaller MP fragments as well as natural (cotton, linen and wool) and synthetic textile fibers may pass through the screens used for sewage treatment. Michielssen et al. (2016) reported that fibers were the predominant form of microliter released from WWTPs and could account for 80% of the material in the final effluent. Confirmation of the composition of the fibers is important when determining the amount of MP material actually discharged to the environment (Ziajahromi et al., 2017b). Talvitie et al. (2017) found that up to 66% of the textile fibers that were released in treated effluent samples were natural and not synthetic.

While effectively treated effluents from WWTP can contain very low concentrations of MPs (0.3×10^{-6} pp L^{-1} (Carr et al., 2016), due to the large volumes of water treated, this can still result in the release of several thousand MP particles to the aquatic environment each day. A WWTP in Detroit, MI (USA) was estimated to discharge approximately 9 billion fibers (natural and synthetic) per day in its effluent, representing the release of 3800 fibers per person per day into the Detroit River (Michielssen et al., 2016). Improved treatment plant designs can reduce this discharge load.

MP fragments and fibers that are retained by the WWTPs usually end up in the sewage sludge. Mahon et al. (2017) found that sludge contained 4200–15,400 MP particles kg^{-1} dry weight. The most common form of MP found in the sludge was fibers (Mahon et al., 2017; Talvitie et al. (2017). Treated sludge, known as biosolids, is often applied to farmland as fertilizers. In the United Kingdom, approximately 80% of sewage sludge is applied to agricultural soils (Horton et al., 2017). Runoff from the fields can resuspend the fibers and transport them to rivers and lakes, or to the ocean.

MP Collection and Identification

MP particles have been found in freshwater ecosystems throughout the world, floating at the water's surface, suspended in the water column, intermixed with beach and bottom sediments, and even in the digestive tracts of aquatic organisms. The methods used to collect MPs in freshwaters are similar to those used in marine environments (Wang and Wang, 2018). MPs can be collected from the water using nets, trawls, pumps and grab samples. The mesh that is used to collect and concentrate the samples varies considerably (20–500 μm), with smaller meshes clogging faster, but retaining smaller sized MPs. Metallic grabs or corers are used to collect samples of bottom sediments, while the top five cm of beach sediments are generally collected by hand and sieved. A variety of techniques are used to collect biota, depending on the study design. The distribution of MPs in freshwaters has not been documented as well as that of marine ecosystems. The diverse hydrological patterns of inland lakes and rivers that receive runoff from urban and agricultural areas leads to large variations in MP concentrations within the water bodies and requires a large number of samples to adequately describe the distribution pattern of the particles (Horton et al., 2018).

MP particles must be sorted from the other material in the water or sediment samples prior to enumeration and identification. Some researchers use density separation, which involves adding solutions with a high specific gravity ($ZnCl_2$, NaCl, NaBr, NaI, $CaCl_2$, $ZnBr_2$, or Na_2WO_4) to the sample and then collecting the MPs that float to the top of the sample. Samples are often sieved to isolate materials of different sizes for analysis. If a lot of plankton is present in the samples, chemicals (KOH, H_2O_2, Fe^{2+}, acids, and enzymes) may be added to digest the organic matter. MPs are then examined qualitatively or quantitatively (Fig. 3). Qualitative analyses involve visually or microscopically sorting and counting particles that resemble plastic materials, while qualitative analyses include additional steps to confirm the identity of the plastic polymers. Microscopy generally cannot differentiate MP particles less than 50 μm (Alam et al., 2019) and it is critical to use quantitative techniques to avoid counting paint chips, natural cellulose fibers or fly ash as MPs (Eriksen et al., 2013). On the other hand, due the large numbers of particles in samples, fluorescent dyes (Maes et al., 2017) and Nile Red (Andrady, 2011) are being used during sorting in qualitative analyses to rapidly distinguish the stained MPs from the background material. However, some synthetic polymers do not stain and this can affect the identification and quantification of MP particles and fibers (Prata et al., 2019a).

Hyper-Spectral Imaging (HSI) in the Short-Wave Infrared (SWIR) range is another technique used to detect plastic by differentiation of absorption bands in the spectra region and vibration of C–H molecular bonds. Serranti et al. (2018) found that this technique can identify plastic debris floating in the ocean (PP, PE and PS) in a size range of 1–5 mm. Elemental analysis using Scanning Electron Microscopy with energy dispersive X-ray spectroscopy (SED/EDX) has been used to differentiate particles with high organic content from plastic (Girão et al., 2016; Wagner et al., 2017).

In order to advance MP research, the correct identification of MP particles is essential and Fourier Transform Infrared (FTIR) and Raman spectroscopies are the main quantitative techniques used to identify synthetic polymers of plastics. These vibration spectroscopy techniques detect the chemical bonds in the samples and can differentiate between the different types of synthetic polymers and natural debris. These instruments use their own libraries to identify the polymers. Micro-FTIR with Attenuated Total Reflection (ATR) and micro-Raman can analyze particles as small as 5–10 μm (Käppler et al., 2016). Pyrolysis-gas chromatography-mass spectrometry (Pyr-GC-MS) is another technique used to obtain information about plastic polymers. This technique is based on thermal degradation and pyrolysis of polymer fragments which are separated by GC and characterized by MS (Mintenig et al., 2018). Thermal extraction desorption gas chromatography mass spectrometry (TED-GC-MS) permits the identification of MPs without requiring any sample treatment (Dümichen et al., 2017).

Table 2 Worldwide distribution of MPs in freshwater biomes.

Year of publication	*Location*	*Matrix analyzed*	*Size range of MPs analyzed (μm)*	*Methods for counting and identifying MPs*	*Main fingerprinting and morphology of MPs*	*Concentration of MPs (plastic particles per unit volume or area)*	*References*
Africa							
2018	Bloukrans River Systems (South Africa)	Sediment	63–5000	Dried, sieved, density separation, filtered, microscopic examination		Summer 6.3 ± 4.3 pp kg^{-1} dw; Winter 160 ± 139 pp kg^{-1} dw	Nel et al. (2018)
2019	Streams near Bizerte Lagoon (Tunisia)	Sediment	200–5000	Dried, density separation, filtered, microscopic examination, FTIR-ATR	98.7% fibers, PP, PE	2340 ±227 to 6920 ± 396 pp kg^{-1} dw	Toumi et al. (2019)
Asia							
2014	Lake Hoysgol (Mongolia)	Surface water	355–4750	Net sample, sieved, digested, density separation, microscopic examination	Fragments, films, lines/fibers	0.020 pp m^{-2}	Free et al. (2014)
2015	Three Gorges Reservoir and tributaries (China)	Surface water	112–5000	Net sample, sieved, settled, supernatant collected, microscopic examination, FTIR-ATR	PE, PP	0.192–13.6 pp m^{-3}	Zhang et al. (2015)
2016	Taihu Lake (China)	Surface water	5–5000	Grab sample, filtered, digested, microscopic examination, μFTIR, SEM/EDS	Fibers, Cellophane, PET, PES, Terephthalic acid, PP	3.4–25.8 pp L^{-1}	Su et al. (2016)
2016	Taihu Lake (China)	Surface water	333–5000	Net sample, filtered, digested, microscopic examination, μFTIR, SEM/EDS	Fibers, Cellophane, PET, PES, Terephthalic acid, PP	0.01–6.8 pp m^{-2}	Su et al. (2016)
2016	Taihu Lake (China)	Sediment	5–5000	Grab sample, density separation, filtered, digested, microscopic examination, μFTIR, SEM/EDS	Fibers and fragments Cellophane, PET, PES, Terephthalic acid, PP	11–234.6 pp kg^{-1} dw	Su et al. (2016)
2016	Tibet Plateau Lakes (China)	Shoreline sediment	1000–5000	Dried, sieved, density separation, filtered, microscopic examination, μRaman, SEM	PE, PP	8 ± 14 to 563 ± 1219 pp m^{-2}	Zhang et al. (2016)
2017	Vembanad Lake (Kerala, India)	Sediment	<5000	Sieved, dried, digested, density separation, filtered, microscopic examination, μRaman	Films, foams common, PE, PS, PP	96–496 pp m^{-2} (252.8 ± 25.8 pp m^{-2})	Sruthy and Ramasamy (2017)
2017a	Urban Lakes and Rivers (China)	Surface water	50–5000	Pump sample, sieved, digested, filtered, microscopic examination, SEM, FTIR	Fibers PET, PP, PE	1660 ± 639.1 to 8925 ± 1591 pp m^{-3}	Wang et al. (2017a)
2017b	Beijiang River (China)	Littoral zone sediment	<5000	Dried, density separation, ultrasound, filtration, microscopic examination, SEM, FTIR	PE, PP	178 ± 69 to 544 ± 107 pp kg^{-1} dw	Wang et al. (2017b)
2018	Three Gorges Reservoir (China)	Surface water	48–5000	Pump sample, sieved, digested, filtered, microscopic examination, μRAMAN, SEM	Fibers PP, PE, PS	1597–12,611 pp m^{-3} (4703 ± 2816 pp m^{-3})	Di and Wang (2018)
2018	Three Gorges Reservoir (China)	Sediment	48–5000	Grab sample, density separation, sieved, digested, filtered, microscopic examination, μRAMAN, SEM	Fibers PS, PP, PE	25–300 pp kg^{-1} ww (82 ± 60 pp kg^{-1} ww)	Di and Wang (2018)
2018	West and South Dongting Lake (China)	Surface water	45–5000	Pump, filtration, digestion, density separation, filtration, microscopic examination, Raman	Fibers PS, PET, PP, PE	367–2317 pp m^{-3}	Jiang et al. (2018)

(*Continued*)

Table 2 (Continued)

Year of publication	*Location*	*Matrix analyzed*	*Size range of MPs analyzed (μm)*	*Methods for counting and identifying MPs*	*Main fingerprinting and morphology of MPs*	*Concentration of MPs (plastic particles per unit volume or area)*	*References*
2018	West and South Dongting Lake (China)	Shoreline sediment	<5000	Dried, density separation, digestion, density separation, filtration, microscopic examination, Raman	Fibers PET, PE, PS, PP, PVC	200–1150 pp kg^{-1} dw	Jiang et al. (2018)
2018	Saigon River (Vietnam)	Surface water	50–4850	Grab samples, digested, density separation, filtered, microscopic examination, FTIR	PES fibers,	172–519 pp L^{-1} (fibers)	Lahens et al. (2018)
2018	Saigon River (Vietnam)	Surface water	50–4850	Net samples, digested, density separation, filtered, microscopic examination, FTIR	PE, PP fragments	10–223 pp m^{-3} (fragments)	Lahens et al. (2018)
2018	Pearl River (China)	Sediment	20–5000	Grab sample, dried, density separation, filtration, digested, microscopic examination, μFTIR	55% fibers, 43% fragments PE, PP	80–9597 pp kg^{-1}	Lin et al. (2018)
2018	Pearl River (China)	Surface water	20–5000	Bulk water sample sieved, digested, density separation, filtered, microscopic examination, μFTIR	80% fibers, 19 % fragments PP, PE, PET	379–7924 pp m^{-3}	Lin et al. (2018)
2018	Rivers near Shanghai (China)	Shoreline sediment	100–5000	Dried, density separation, filtered, microscopic examination, μFTIR	89% spheres, 8% fibers, 4% fragments, PP, PES, rayon	802 ± 594 pp kg^{-1} dw	Peng et al. (2018)
2018	Taihu Lake, (China)	Surface water	20–5000	Bulk water sample, sieved, digested, filtered, microscopic examination, μFTIR	Fibers	0.5–3.1 pp L^{-1}	Su et al. (2018)
2018	Taihu Lake, (China)	Sediment	20–5000	Dried, density separation, sieved, digested, filtered, microscopic examination, μFTIR	Fibers	15–160 pp kg^{-1}	Su et al. (2018)
2018	Dongting and Hong Lakes (China)	Surface water	50–5000	Bulk water sample, sieved, digested, filtered, microscopic examination, SEM, μRaman	Fibers, PE, PP	900–4650 pp m^{-3}	Wang et al. (2018)
2019	Ciwalengke River (Indonesia)	Surface water	50–2000	Grab sample, filtration, density separation, filtration, microscopic examination, Raman	65% fibers, PES, nylon	5.85 ± 3.28 pp L^{-1}	Alam et al. (2019)
2019	Ciwalengke River (Indonesia)	Sediment	50–2000	Density separation, filtration, microscopic examination, Raman	91% fibers, PES, nylon	30.3 ± 15.9 pp kg^{-1} dw	Alam et al. (2019)
2019	Selenga River Basin (Mongolia)	Shoreline debris	<5000	Visual collection, digestion, filtration, microscopic examination, μFTIR	99% PS foam	1.20 ± 1.21 pp m^{-2}	Battulga et al. (2019)
2019	Rivers, Tibet Plateau (China)	Surface water	45–5000	Pump, filtration, digestion, density separation, filtration, microscopic examination, Raman	69–93% fibers, PE	483–967 pp m^{-3}	Jiang et al. (2019)
2019	Rivers, Tibet Plateau (China)	Shoreline sediment	<2000	Dried, sieved, density separation, digestion, density separation, filtration, microscopic examination, Raman	54–81% fibers, PET	50–195 pp kg^{-1} dw	Jiang et al. (2019)
2019	Rice and fish culture ponds (China)	Water	20–5000	Filtration, digestion, microscopic exam, μFTIR	Fibers, films PP and PE	0.4 ± 0.1 pp L^{-1}	Lv et al. (2019)
2019	Rice and fish culture ponds (China)	Sediment	20–5000	Density separation, filtration, digestion, microscopic exam, μFTIR	Fibers, PE, PP, PVC	10.3 ± 2.2 pp kg^{-1}	Lv et al. (2019)

2019	Feilaixia Reservoir (China)	Surface water	112–5000	Net sample, sieved, filtered, dried, smaller particles microscopic examination, larger particles picked out for μFTIR	Foams, films, fragments and fibers PP and PE	0.56 ± 0.45 pp m^{-3}	Tan et al. (2019)
2019	Lake Ulansuhai (China)	Surface water	48–5000	Pump, sieved, digestion, filtration, microscopic examination, FTIR, SEM, EDS	Fibers PE, PST, PET	1760 ± 710 to $10{,}120 \pm 4090$ pp m^{-3}	Wang et al. (2019b)
2019	Urban Lakes, Changsha (China)	Surface water	50–5000	Filtered, digested, microscopic inspection, SEM, Raman	PP, PE, PET, PA fibers (lines), also PP, PE, PS fragments, films, foam	2425 ± 248 to 7050 ± 1061 pp m^{-3}	Yin et al. (2019)
2019	Yongjiang River (China)	Surface water	50–5000	Pump, sieved, digestion, filtration, microscopic examination, Raman	73–92% fibers, 14% fragments, PET, nylon, PE	2345 ± 1858 pp m^{-3} (500–7700)	Zhang et al. (2019b)
2019	Yongjiang River (China)	Sediment	48–5000	Density separation, filtration, digestion, microscopic examination, Raman	60% fibers, 30% fragments, PE, PP, PET	285 ± 110 pp kg^{-1} ww (90–550)	Zhang et al. (2019b)
Australia							
2019	Urban wetlands (Melbourne)	Sediment	35–1000	Sieved, dried, density separation, filtered, microscopic examination	Fragments, beads, fibers	2–147 pp kg^{-1} dw	Townsend et al. (2019)
Europe							
2012	Lake Geneva (Switzerland)	Surface water	300–5000	Net sample, sieved, dried, microscopic examination	Primary and secondary particles	0.048 pp m^{-2}	Faure et al. (2012)
2013	Lake Garda (Italy)	Beach Sediment	<5000	Density separation, microscopic examination, Raman	Fragments 46% PS, 43% PE, 10% PP	108–1108 pp m^{-2}	Imhof et al. (2013)
2015	River Seine (France)	Surface water	100–5000	Net sample, filtered, microscopic examination	Fibers	3–108 pp m^{-3}	Dris et al. (2015)
2015	River Seine (France)	Surface water	330–5000	Net sample, filtered, microscopic examination	Fibers, fragments, spheres	0.28–0.47 pp m^{-3}	Dris et al. (2015)
2015	6 Lakes (Switzerland)	Surface waters	300–5000	Net sample, sieved, digestion, microscopic examination, FTIR	Fragments, films, foams PE, PP, PS	0.091 pp m^{-2} (0.026 mg m^{-2})	Faure et al. (2015)
2015	5 Rivers (Switzerland)	Surface waters	300–5000	Net sample, sieved, digestion, microscopic examination, FTIR	Fragments, foams PE, PP, PS	7 ± 0.2 pp m^{-3} (1.4 ± 3.4 mg m^{-3})	Faure et al. (2015)
2015	6 Lakes (Switzerland)	Beach sediment	300–5000	Density separation, sieved, microscopic examination, smaller fraction digested before examination, FTIR	Foams, fragments PE, PP, PS	1300 ± 2000 pp m^{-2} (920 ± 1500 mg m^{-2})	Faure et al. (2015)
2015	River Rhine (Germany)	Beach sediment	63–5000	Dried, sieved, density separation, digestion, filtration, microscopic examination, FTIR	Fragments, spheres PE, PP, PS	228–3763 pp kg^{-1} dw (21.8–932 mg kg^{-1} dw)	Klein et al. (2015)
2015	River Main (Germany)	Beach sediment	63–5000	Dried, sieved, density separation, digestion, filtration, microscopic examination, FTIR	Fragments, spheres PE, PP, PS	786–1368 pp kg^{-1} dw 43.5–459 mg kg^{-1} dw	Klein et al. (2015)
2015	Rhine River (Basel to Rotterdam)	Surface water	300–5000	Net samples, sieved, digestion, sieved, density separation, microscopic examination, FTIR	Spheres, fragments PS, PP	0.89 pp m^{-2}	Mani et al. (2015)
2016	Lakes Bolsena and Chiusi (Italy)	Beach sediment	300–5000	Sieved, dried, density separation, digestion, dyed, filtered, UV microscopic examination, SEM	Fragments, fibers	1922–2117 pp m^{-2} (112–234 pp kg^{-1} dw)	Fischer et al. (2016)
2016	Lakes Bolsena and Chiusi (Italy)	Surface water	300–5000	Net sample, sieved, density separation, dyed, filtered, UV microscopic examination, SEM	Fragments, fibers	0.8–4.4 pp m^{-3}	Fischer et al. (2016)
2017	River Thames (United Kingdom)	Sediment	1000–4000	Sieved, dried, microscopic examination, density separation, filtration, dried, microscopic examination, Raman	Fibers, fragments	185 ± 42 to 660 ± 77 pp kg^{-1} dw	Horton et al. (2017)

(*Continued*)

Table 2 (Continued)

Year of publication	*Location*	*Matrix analyzed*	*Size range of MPs analyzed (μm)*	*Methods for counting and identifying MPs*	*Main fingerprinting and morphology of MPs*	*Concentration of MPs (plastic particles per unit volume or area)*	*References*
2018	Rivers Marne and Seine (France)	Surface water	>80	Net sample, digestion, density separation, filtered, microscopic examination, FTIR	Fibers rayon, PET, PP, PA	Site averages of 22.1 ± 25.3 to 100.6 ± 99.9 pp m^{-3}	Dris et al. (2018)
2018	Mersey and Irwell Rivers (England)	Sediment	63–5000	Cylindrical sampler, sieving, dried, density separation, filtration, microscopic examination, FTIR	Fibers, fragments, beads	Pre flood average 6350 pp kg^{-1}; post flood average 2812 pp kg^{-1}	Hurley et al. (2018)
2018	Antua River Basin (Portugal)	Shoreline sediment	55–5000	Grab sample, dried, sieved, digested, density separation, filtration, microscopic examination, FTIR	PE, PP, PS, PET Fragments, fibers, films	18–629 pp kg^{-1} dw; 2.6–71.4 mg kg^{-1} dw	Rodrigues et al. (2018)
2018	Antua River Basin (Portugal)	Surface and bottom water	55–5000	Pump sample, sieved, digested, density separation, filtration, microscopic examination, FTIR	PE, PP, PS, PET Fragments, films, foams, fibers	58–1265 pp m^{-3}; 5–51.7 mg m^{-3}	Rodrigues et al. (2018)
2018	Teltow Canal (Germany)	Surface water	450–5000	Grab sample, sieved, dried, digested, filtered, SWIR imaging, microscopic check	PE, PP	0.01–95.8 pp L^{-1}	Schmidt et al. (2018)
2018	Subalpine Lakes (Italy)	Surface water	1000–5000	Net sample, sieved, digested, microscopic examination, FTIR	74% Fragments, PE, EPS, PP	0.004–0.057 pp m^{-2}	Sighicelli et al. (2018)
2018	River Tame and Tributaries (United Kingdom)	Sediment	63–4000	Wet sieved, dried, density separation, microscopic examination, FTIR	49% Fragments 22% fibers PE, PVC, PMMA	110 kg^{-1} dw	Tibbetts et al. (2018)
2019	River Kelvin (Scotland)	Shoreline sediment	11–2800	Dried, sieved, density separation, filtration, microscopic examination, SEM- EDS	88–95% fibers	161–432 pp kg^{-1} dw	Blair (2019)
2019	Rivers, reservoirs and fish ponds (Hungary)	Surface water	100–2000	Pump sample, sieved, density separation, digested, filtration, μFTIR-ATR	PP, PE, PES, PS,	3.52–32.05 pp m^{-3}	Bordós et al. (2019)
2019	Rivers, reservoirs and fish ponds (Hungary)	Sediments		Grab sample, density separation, digestion, filtration, μFTIR-ATR	PP, PS, PES	0.46–1.62 pp kg^{-1} 0.8 ± 0.37 pp kg^{-1}	Bordós et al. (2019)
2019	Stormwater retention ponds (Denmark)	Surface water	10–2000	Pump sample, filtration, digestion, density separation, FTIR imaging	17 polymers, mainly PP	490–23,000 pp m^{-3} (85–1143 μg m^{-3})	Liu et al. (2019)
2019	Viborg stormwater retention pond (Denmark)	Surface water	10–500	Grab sample, filtered, digested, FPA-μFTIR imaging	PEST, PP, Acrylic	270 pp L^{-1} (4.2 μg L^{-1})	Olesen et al. (2019)
2019	Viborg stormwater retention pond (Denmark)	Sediment	10–500	Core sample, sieved, dried, density separation, digestion, FPA- μFTIR imaging	PP	950,000 pp kg^{-1} dw (0.4 g kg^{-1} dw)	Olesen et al. (2019)
North America							
2011	California Rivers (USA)	Surface water	1000–4750	Net tow, sieved, microscopic examination	Foam, pellets, fragments, films, lines	<1–153 pp m^{-3}	Moore et al. (2011)

2011	Lake Huron (Canada)	Beach sample	Pellets <5000 Fragments >5000	Visual collection, sonication, μFTIR- ATR, SEM	PE pellets, PP fragments, PS foams	0–408 pp m^{-2}; mean 37.8 pp m^{-2}	Zbyszewski and Corcoran (2011)
2013	Laurentian Great Lakes (USA)	Surface water	355–4750	Net samples, density separation, sieved, microscopic examination and SEM/EDS of smaller particles	Pellets, fragments	0–0.466 pp m^{-2}; average 0.43 pp m^{-2}	Eriksen et al. (2013)
2014	St. Lawrence River (Canada)	Sediment	>500	Grab sample, sieved, microscopic examination, scanning calorimeter	Microbeads	0–136,926 pp m^{-2}; 13,832 ± 13,677 pp m^{-2}	Castaneda et al. (2014)
2014	Lakes Erie and St Clair (Canada/ USA)	Beach sample	<2000	Visual collection, cleaned, dried, hand sorted, FTIR, SEM	Fragments, foams PE and PP	0.18–8.38 pp m^{-2}	Zbyszewski et al. (2014)
2015	Lake Ontario (Canada)	Beach sediment	<1000, 1000–5000, >5000	Visual collection, sorting, enumeration, Raman	PS foam, industrial pellets, fragments	21.8 pp m^{-2} (excluding PS)	Corcoran et al. (2015)
2015	Lake Ontario (Canada)	Bottom sediment	<0.500, 500–710, 710–850, 850–1000, >1000	Core sample, dried, sieved, density separation, microscopic examination, FTIR	74% PE	0.087–0.62 pp g^{-1} dw	Corcoran et al. (2015)
2016	Great Lakes Tributaries (USA)	Surface water	333–4750	Net samples, sieved, digested, filtered, microscopic examination	Fibers, fragments	0.05–32 pp m^{-3}	Baldwin et al. (2016)
2016	Lake Ontario	Nearshore sediment	>63	Grab, core or sediment trap samples, dried, sieved, sorted, density separation of smaller fraction, microscopic examination, Raman, FTIR	Fibers, fragments	980 pp kg^{-1} dw	Ballent et al. (2016)
2016	Lake Ontario	Tributary sediment	>63	Grab samples, dried, sieved, sorted, density separation of smaller fraction, microscopic examination, Raman, FTIR	Fragments, fibers	610 pp kg^{-1} dw	Ballent et al. (2016)
2016	Lake Ontario	Beach sediment	>63	Core samples, dried, sieved, sorted, density separation of smaller fraction, microscopic examination, Raman, FTIR	Fragments, fibers	140 pp kg^{-1} dw	Ballent et al. (2016)
2016	Raritan River (USA)	Surface water	125–2000	Net tows, sieved, dried, digested, density separation, microscopic examination,		Upstream WWTP 24 ± 11.4 pp m^{-3} Downstream WWTP 71.7 ± 60.2 pp m^{-3}	Estahbanati and Fahrenfeld (2016)
2016b	Lake Michigan (USA)	Surface water	355–999 1000–4749 >4750	Net tow, sieved, digested, filtered, microscopic examination, SEM/EDS,	79% fragments, 14% fibers	Average 0.017 pp m^{-2}	Mason et al. (2016b)
2016	North Shore Channel (USA)	Surface water	333–2000	Net tows, sieved, dried, digested, density separation, filtered, microscopic examination	Fibers, fragments	1.94–17.93 pp m^{-3}	McCormick et al. (2016)
2017	Lake Winnipeg, (Canada)	Surface water	333–5000	Net sample, sieved, digested, filtered, microscopic examination, SEM	Fibers	0.053–0.75 pp m^{-2}	Anderson et al. (2016)
2017	Waucana Creek (Canada)	Surface water	80–4750	Net tow, sieved, digested, microscopic examination, hot needle	Fibers, fragments	0.9–7.7 pp m^{-3}	Campbell et al. (2017)
2017	Hudson River (USA)	Surface water	>100	Grab sample, filtered, microscopic examination, μFTIR	Fibers 43% cotton, 22% PET, 22% fluoro–polymer/Teflon, 7% PP, 7% nitrocellulose /clay	Median 0.98 pp L^{-1} (fibers) Estimate 50% plastic	Miller et al. (2017)

(*Continued*)

Table 2 (Continued)

Year of publication	*Location*	*Matrix analyzed*	*Size range of MPs analyzed (μm)*	*Methods for counting and identifying MPs*	*Main fingerprinting and morphology of MPs*	*Concentration of MPs (plastic particles per unit volume or area)*	*References*
2017	Ottawa River (Canada)	Sediment	300–5000	Grab sample, dried, sieved, digested, density separation, microscopic examination	Fibers	220 pp kg^{-1} dw	Vermaire et al. (2017)
2017	Ottawa River (Canada)	Surface water	100–5000	Grab sample, sieved, digested, filtered, microscopic examination	Fibers	0.1 L^{-1}	Vermaire et al. (2017)
2017	Ottawa River (Canada)	Surface water	100–5000	Net sample, digested, filtered, microscopic examination	Fibers, some fragments, beads	1.35 pp m^{-3}	Vermaire et al. (2017)
2018	Gallatin River Watershed (USA)	Surface water	100–9600	Grab samples, filtered, dried, microscopic exam, hot needle, FTIR	80% fibers, 19.7% fragments, PE, PET, rayon	57% samples had pp, average 1.2 pp L^{-1}	Barrows et al. (2018)
2018	Lake Superior (USA)	Surface water	333–5000	Net tows, sieved, digested, density separation, filtration, microscopic examination, Pyr-GC/MS, ATP-FTIR	Fibers, fragments, films PVC, PP, PE, PET	0.037 ± 0.027 pp m^{-2} 0.0012 mg m^{-2}	Hendrickson et al. (2018)
2018	Snake and Columbia Rivers (USA)	Surface water	100–5000	Grab samples, filtered, dried, microscopic examination, hot needle test, Raman	58% fibers	75% samples had pp, 0.91 ± 1.14 pp L^{-1}	Kapp and Yeatman (2018)
2018	Snake and Columbia Rivers (USA)	Surface water	100–5000	Net samples, filtered, digested, density separation, microscopic examination, hot needle test, Raman	48% fibers	92% samples had pp, 2.57 ± 2.95 pp m^{-3}	Kapp and Yeatman (2018)
2019	Atoyac River Basin (Mexico)	Sediment		Trowel or grab samples, dried, digested, density separation, filtration, microscopic examination, SEM/EDX	26% films, 22% fragment 15% fibers, 11% pellets	4500 ± 702 pp kg^{-1} dw	Shruti et al. (2019)
South America							
2017	Sebutal Lake (Argentina)	Beach sediment	350–5000	Dried, sieved, digested, density separation, microscopic examination, FTIR	Fragments, fibers, foams	704 pp m^{-2}	Blettler et al. (2017)
2019	Pantanal wetlands and rivers (Brazil)	Surface water	68–5000	Net tow, filtered, microscopic examination	50% fibers,19% fragments, 22% pellets, 9% foam	9.6 ± 8.3 pp 100 L^{-1}	de Faria et al. (2019)

Data sources: See References.

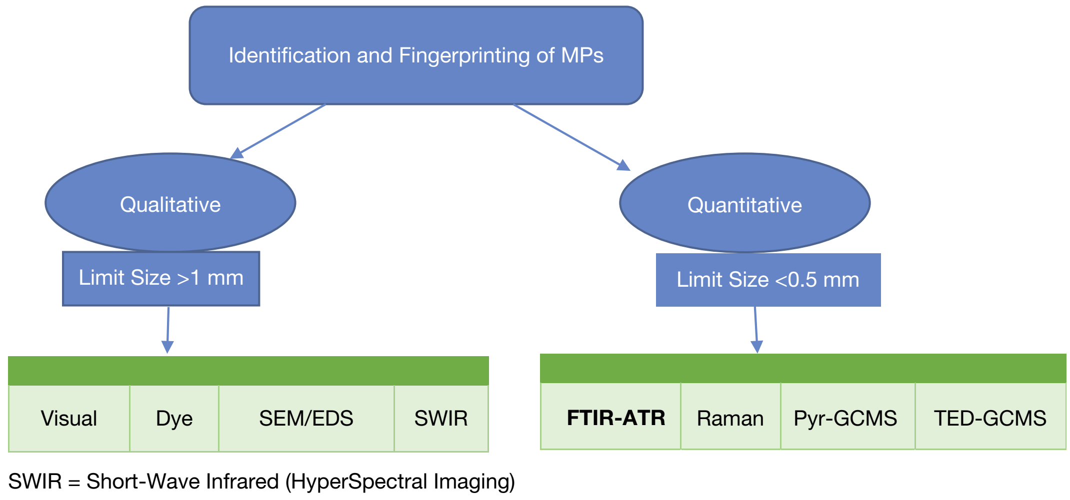

Fig. 3 Analytical techniques to identify MPs in freshwater environmental samples.

Distribution of MPs in Freshwaters

The number of reports of MPs from freshwater lakes and rivers has increased dramatically from 2012 through 2019 (Table 3). Research has been concentrated in Europe, Asia, and North America, but studies of the inland waters of Africa, South America, and Australia have begun in the last few years. The data from various authors is not directly comparable due to the use of different collection and processing methods and a lack of standardization of reporting units (Rios Mendoza and Balcer, 2019a). MP concentrations in surface waters ranged from 0–270 pp L^{-1} by volume and 0–563 pp m^{-2} by surface area. In bottom sediments the concentrations range from 0–10 pp g^{-1} sediment, while on beaches densities varied from 0–0.8 pp g^{-1} or 0–2117 pp m^{-2} (Table 3).

In spite of the lack of harmonization of sampling techniques, certain trends are evident. China has the highest plastic production in the world (Plastics Europe, 2019), including textiles, and makes extensive use of plastic films as mulch for agricultural use (Lv et al., 2019; Wang et al., 2019b). Surface waters of lakes and rivers in China contain some of the highest MP concentrations, often over 1000 pp m^{-3} (Table 3). Worldwide, urban areas generally contain higher concentrations of MPs than rural areas (Table 3), and stormwater retention ponds tend to concentrate MPs, with levels reaching 23–270 pp L^{-1} in water and 9.5×10^5 pp kg^{-1} dw sediment (Liu et al., 2019; Olesen et al., 2019).

With regards to MP morphology, the most common forms were fibers > fragments > pellets or microbeads. The presence of sewage plants increases the concentration of MPs in the water, especially of fibers (Browne et al., 2011; McCormick et al., 2014; Wang et al., 2017a, b).

MPs and Toxic Compounds

MPs contain a mixture of chemicals that are introduced during their manufacture, including additives, stabilizers, flame-retardants, antibacterial and antioxidant agents, pigments, fillers, and plasticizers including phthalates (Hahladakis et al., 2018). These chemicals can be released when MPs enter aquatic environments, leading to the suggestion that plastic be classified as a hazardous material (Rochman et al., 2013a). While information on the interaction of toxic compounds with MPs in freshwater systems is limited at this time, it is an important factor in understanding the adverse effects of environmental pollution on aquatic organisms.

Macro, micro and nanoplastic debris can concentrate and transport toxic compounds in the environment (Mato et al., 2001; Rios et al., 2007) and assessing the abundance of the MP particles and measuring their sizes is essential to determining the real risk of these plastic particles. Smaller particles (Fig. 2) have larger surface areas and can concentrate higher amounts of toxic compounds on their surfaces. MPs adsorb persistent organic pollutants (POPs), semipersistent polycyclic aromatic hydrocarbons (PAHs), heavy metals and pharmaceutical chemicals from the aquatic environment.

The main mechanisms for chemical adsorption on plastics are through hydrophobic (Wang et al., 2019a; Guo and Wang, 2019), electrostatic (Guo and Wang, 2019), van der Waals (Xu et al., 2018), or double bonds interactions (Hüffer and Hofmann, 2016). Rios and collaborators have studied the toxic compounds adsorbed on the surfaces of MPs collected from the Laurentian Great Lakes. They detected PAH concentrations ranging from 0.47–20.26 μg g^{-1} on MPs from the Great Lakes and from the St Louis River Estuary (SLRE). PCBs ranged from nondetectable to 9.86 μg g^{-1} in samples from the Great Lakes and from 0.40–3.46 ng g^{-1} in SLRE. Organochlorine pesticide concentrations varied from 4.3–63.9 ng g^{-1} (Rios Mendoza and Balcer, 2019b; Rios Mendoza, 2019). Faure et al. (2015) reported toxic compounds on MPs from Swiss Lakes with PAHs ranging from 0.086–5.714 μg g^{-1}, PCBs from 0.4–548 ng g^{-1}, and organochlorine pesticides from 1.4–2715 ng g^{-1}.

Table 3 Microplastic concentrations in Wastewater Treatment Plants and efficiency of removal.

Year of publication	*Location*	*Methods for counting and identifying MPs*	*Size range of MPs analyzed (μm)*	*Concentration of MPs (plastic particles per unit volume or area)*	*Percent Removal*	*Dominant particle morphology and polymer*	*References*
2016	8 facilities Los Angeles (USA)	Filtered, microscopic examination, FTIR	45–400	Secondary effluent 1.76 $\times 10^{-5}$ pp L^{-1} Tertiary effluent 0.3–2.4 $\times 10^{-6}$ pp L^{-1}	99%	PE fragments	Carr et al. (2016)
2016a	17 facilities USA	Filtered, microscopic examination	125–>355	Effluents 0.004–0.195 pp L^{-1} Avg. 0.050 pp L^{-1}		59% fibers and 33% fragments	Mason et al. (2016a)
2016	1 facility River Clyde Glasgow, Scotland	Filtered, microscopic examination, FTIR	>65	Influent 15.7 pp L^{-1} Effluent 0.25 pp L^{-1}	98%	67% flakes and 18% fibers in effluent. 28% PE, 20% PA, 12% PP, 12% Acrylic	Murphy et al. (2016)
2017	7 facilities Rhine and Meuse Rivers, Netherlands	Grab sample, filtration, density separation, microscopic examination, μFTIR	10–5000	Influent 68–910 pp L^{-1} Effluent 5–81 pp L^{-1} Sludge 510–760 pp kg^{-1} ww	72%	Fibers dominant	Leslie et al. (2017)
2017	7 facilities Ireland	Grab sample, elutriation, density separation, filtration, microscopic examination, FTIR	<45–250	Sludge 4196–15,385 pp kg^{-1} dw		78% fiber, 7 polymers	Mahon et al. (2017)
2017	12 facilities Oldenburg-East-Frisian (Germany)	Filtration, digestion, density separation, microscopic examination, μFTIR	20–>500	Effluent 0–50 pp >500 μm m^{-3} 10–9000 pp <500 μm m^{-3} Sludge 1000–24000 pp kg^{-1} dw	97%	14 polymers, PE fibers dominant	Mintenig et al. (2017)
2017b	3 facilities Sydney, Australia	Filtration, digestion, density separation, stained, microscopic examination, FTIR-ATR	25–500	Tertiary treatment effluent 0.28 pp L^{-1}	90–98%	PET fibers	Ziajahromi et al. (2017b)
2018	2 facilities Turkey	Filtered, digested, density separation, microscopic examination, μRaman	55–5000	Influent 12,000–36,000 pp m^{-3} Effluent 2700–8700 pp m^{-3}	73–79%	60% fibers 7 polymers, 51% PET, 29% PE, 14% PP	Gündoğdu et al. (2018)
2018	1 facility Mikkeli (Finland)	Grab sample, filtered, digested, digital optical microscopy, μFTIR, μRaman	250–5000	Influent 57.6 pp L^{-1} Effluent 1.0 pp L^{-1} Sludge 8.2 – 301.4 pp g^{-1} dw	98.4%	79% PET fibers, 4% Nylon fibers, 11% PE particles	Lares et al. (2018)
2018	10 facilities Denmark	Grab sample, filtered digested, Focal Plane Array (FPA) and FTIR imaging	10–500	Median Influent 7216 pp L^{-1} Effluent 54 pp L^{-1}		Mainly particles/ fragments, PES, PE, PP, acrylates	Simon et al. (2018)

Data sources: See References.

Interactions of Aquatic Organisms with MPs

MPs as a Habitat for Microbes

Plastic debris can be colonized by a variety of microbes and develop into a unique aquatic community known as the "plastisphere" (Zettler et al., 2013). These diverse floating colonies can be transported long distances across the water surface. As the biofilm grows and increases in density, the plastic material and its associated microbial community can sink, thus serving as a vector for the spread of potentially harmful bacteria and other organisms throughout the water column (McCormick et al., 2014).

McCormick et al. (2014) found that microplastic pellets and fragments collected near a WWTP in the North Shore Channel of Chicago IL developed an extensive biofilm of prokaryotic organisms with a density up to 0.3 cells μm^{-2} of surface area. DNA sequencing revealed that the bacterial community on the MP particles was distinct from that of the water column or on natural organic material from the same area and contained a high abundance of *Pseudomonas*, which may be able to degrade MPs. Several genera of bacteria that contain pathogenic strains were also found at elevated levels on MP particles (McCormick et al., 2014).

By serving as a substrate, MPs enable bacteria to grow in close contact with each other, potentially allowing for increased horizontal gene transfer (HGT) or exchange of genetic information via plasmid transfer. This exchange could lead to the formation of more antibiotic resistant strains of bacteria. Arias-Andres et al. (2018) studied this hypothesis using a strain of *Escherichia coli* with a plasmid that encodes for trimethoprim resistance. They found a large increase in the HGT rate in bacteria grown in water from Lake Stechlin, Germany that contained microplastic particles versus controls without microplastic.

Field Studies on Ingestion and Other Interactions

MP particles that enter freshwater environments may be consumed by a variety of aquatic organisms including clams, worms, crayfish, insects, tadpoles, and fish (Table 4). The ability to detect small MPs in aquatic organisms is affected by the analytical procedures that are used to extract and identify the possible MP particles. Some researchers dissected the digestive tracts and examined the contents microscopically, while others chemically dissolved the digestive tract or the entire organism to remove organic material and then separated the MP particles by flotation or filtration prior to microscopic examination.

Recent studies have included the use of FTIR spectroscopy to confirm that the microparticles are actually MPs and not other natural materials such as cellulose, cotton, wood, or sand grains. The most commonly ingested form of MPs was fibers comprised of PES, PE/PET, PE, and PP (Table 4). The percentage of examined organisms that contained MPs varied considerably (Table 4), ranging from 20% or less (Sanchez et al., 2014; Faure et al., 2015; Phillips and Bonner, 2015; Biginagwa et al., 2016; Roch and Brinker, 2017; Roch et al., 2019; Su et al., 2019) to over 75% (Jabeen et al., 2017; Silva-Cavalcanti et al., 2017; Campbell et al., 2017; Nel et al., 2018; Xiong et al., 2018). While MP particles were commonly found in aquatic organisms, the density of MP per individual was generally quite low (<3 pp ind^{-1}, Table 4). Higher densities of MPs were found in bottom feeding fish and those from urban locations with more environmental contamination (Zheng et al., 2019; Silva-Cavalcanti et al., 2017; Peters and Bratton, 2016; Sanchez et al., 2014; Phillips and Bonner, 2015).

While field studies have concentrated on determining the number of MPs ingested by aquatic animals, organisms can interact with plastic debris in other ways. Caddisfly larvae build external cases or tubes from sand and other materials that they find in the environment (Fig. 4) and can incorporate MPs if they are present. MP particles were found in 7% of the cases of caddisfly larva collected from an urban section of the River Tame, United Kingdom (Tibbetts et al., 2018) and in 58% of the cases of the species *Lepidostoma basale* from a stream in Germany (Ehlers et al., 2019).

Laboratory Studies

Several laboratory experiments have investigated the interactions of MPs with aquatic organisms ranging from microbes to fish under more controlled conditions (Table 5). Organisms have been exposed to MPs via water, sediment, or through their diets. The test concentrations of MPs have been reported using a variety of units, including number or weight of plastic particles per unit weight of sediment, or per volume of test medium. Due to the wide variation in size and weights of the MP particles, some authors have begun to report test concentrations both by weight and number of particles. Experimental exposures of MP concentrations have ranged from 0.5–2570 pp g^{-1} or 2–20 mg g^{-1} dry weight in sediment, and from $50–1 \times 10^{10}$ pp L^{-1} or $0.01–1 \times 10^4$ mg L^{-1} in water. The most commonly tested forms of MP have been PE and PS spheres and fragments and PES/PET fibers (Table 5).

Initial studies were designed to determine whether or not MPs of various shapes and sizes could be ingested and to what degree the particles would accumulate in the guts of the test organisms (Imhof et al., 2013; Rochman et al., 2017; Murphy and Quinn, 2018; Li et al., 2019; Cuthbert et al., 2019). While plastic particles are readily ingested by a variety of organisms (Table 5), they are also egested fairly rapidly, resulting in a low body burden in most test organisms (<15 pp ind^{-1}, or <4.1 pp g^{-1}). Some size selectivity was observed, with *Gammarus* ingesting smaller MP fragments (32–63 μm) at the highest rates (Straub et al., 2017; Weber et al., 2018). Li et al. (2019) found that the number of fibers ingested by Asian clams was related to the flexibility of the plastic polymer (Table 5), with PET fibers favored over PEA and AC. The more rigid PA, RA and PVA fibers were not ingested as readily.

Ingested MP materials can be retained in an organism's digestive tract or passed through the body to be egested in the feces. Smaller sized particles may move from the gut into other body tissues by crossing the epithelial lining. To determine the concentration of MP in various parts of an organism, researchers may dissect or dissolve the specific organs to extract and count the individual plastic particles. When exposing organisms to smaller particles, the practice has been to use MPs labeled with fluorescent dyes and then use imaging to detect the dye and particles in various locations in the organism. The strength of the fluorescent signal is correlated to the number of particles ingested. Organisms can be damaged during sample preparation (Guilhermino et al., 2018), allowing particles to appear in other tissues besides the gut. While MPs have primarily been found in the digestive tracts of organisms (Table 5), small MPs (1–5 μm) have been reported from various tissues of Asian clams (Guilhermino et al., 2018), zebra mussels (Magni et al., 2018), *Daphnia* (Rosenkranz et al., 2009) and zebrafish (Lu et al., 2016). Schür et al. (2019) questioned the ability of fluorescent microbeads to move from the gut of *Daphnia* though the peritrophic membrane (which coats the food pellets in cladocerans), and then through the gut epithelium into lipid droplets. They found that fluorescent dyes can leach from plastic particles and end up in lipids, even when MP particles were not visible. Catarino et al. (2019)

Table 4 Ingestion of MPs by freshwater organisms in natural environments.

Organism	*Location*	*Processing procedure*	*% Organisms with MPs*	*Average number of pp ind^{-1} or weight^{-1} of organism*[a]	*Dominant particle morphology and polymer*	*References*
Asian clam *Corbicula fluminea*	Taihu Lake, China	Soft tissue extracted, digested, filtered 100 μm mesh, microscopic examination, FTIR or SEM/EDS		0.2–12.5 pp g^{-1} ww	Fibers	Su et al. (2016)
Asian clam *Corbicula fluminea*	Yangtze River Basin, China	Soft tissue extracted, digested, filtered, microscopic examination, FTIR		0.4–5.0 pp ind^{-1} 0.3–4.9 pp g^{-1} ww	Fibers	Su et al. (2018)
Annelid worm *Tubifex tubifex*	Salford Quays, Manchester, UK	Organisms rinsed, 24 h depuration, tissue digested, microscopic examination, hot needle test, FTIR	48	0.8 ± 1.01 pp ind^{-1} 129 ± 65.4 pp g^{-1} ww	87% PE/PET, AC, and PP fibers 14% PST, PE, PP fragments	Hurley et al. (2017)
Crayfish *Procambarus clarkii*	Rice and fish culture ponds, China	Digestive tract removed, digested, filtered, microscopic examination, FTIR		2.5 ± 0.6 pp ind^{-1}	Fibers PE and PP	Lv et al. (2019)
Mayflies Heptageniidae and Baetidae Caddisflies Hydropsychidae	Rivers in South Wales, UK	Field collection, half preserved, half allowed to depurate. Organisms rinsed, homogenized, digested, microscopic examination, dark field spectroscopy for particles from 0.5–5 mm	≅50	Maximum 0.14 pp mg^{-1} dw Depurated organisms ≅ 0.65 pp ind^{-1} preserved specimens 1.15 pp ind^{-1}		Windsor et al. (2019)
Chironomid larvae *Chironomus* sp.	Bloukrans River System, South Africa	Collected, preserved, weighed, digested, filtered, microscopic examination	75–98	0.37 ± 0.44 pp mg^{-1} ww summer 1.12 ± 1.19 pp mg^{-1} ww winter		Nel et al. (2018)
Tadpoles	Small waterbodies, Yangtze River Delta, China	Samples pooled, digested, filtered, microscopic examination, μFTIR-ATR		0–2.73 pp ind^{-1} 0–168 pp g^{-1} ww	68% PES fibers, 7% PP fibers	Hu et al. (2018)
Fish Bleak *Alburnus alburnus*, Common dace *Leuciscus leuciscus* European perch *Perca fluviatilis*, Roach *Rutilus rutilus*	Lake Geneva, Switzerland	Preserved specimens, GI tract dissected and examined microscopically	7.5	1 Bleak had 31 fibers 2 Dace contained fragments No items in other species		Faure et al. (2015)
Fish—Gudgeon *Gobio gobio*	Rivers in France	GI tract dissected, microscopic examination,	12 (range 9.5–42% by site)			Sanchez et al. (2014)
Fish—Composite sample of 44 species from 12 families	Rivers in Texas near Gulf of Mexico, USA	Stomach and upper GI tract dissected, microscopic exam, FTIR	8.2 (range 5–29% by drainage)		1.3% had filaments 2.7% fragments and 3.1% films PP, PES, PA, PS, nylon	Phillips and Bonner (2015)
Fish—Bluegill, *Lepomis macrochirus* Longear sunfish, *L. megalotis*	Brazos River Basin, Texas, USA	Stomach contents removed, washed through filters, microscopic exam	45 44	0.34–1.33 pp ind^{-1} of different size classes	96% threads	Peters and Bratton (2016)
Fish–Nile Perch *Lates niloticus* Nile Tilapia *Oreochromis niloticus*	Lake Victoria, Tanzania	Entire GI tract removed, digested, 250 μm sieve, microscopic examination, FTIR	20 20		PE, PU, PES, PE/PP, silicone rubber	Biginagwa et al. (2016)

Fish—6 species	Taihu Lake, China	Entire GI tract removed, digested, floatation, 5 μm filter, microscopic examination, FTIR.	95.7 with MPs (43.5% also contained mesoplastics[b])		57–88% fibers	Jabeen et al. (2017)
Cyprinus carpio				2.5 ± 1.3 pp ind^{-1}		
Carassius auratus				1.9 ± 1.0 pp ind^{-1}		
Hypophthalmichthys molitrix				3.8 ± 2.0 pp ind^{-1}		
Pseudorasbora parva				2.5 ± 1.8 pp ind^{-1}		
Megalobrama amblycephala				1.8 ± 1.7 pp ind^{-1}		
Hemiculter bleekeri				2.1 ± 1.1 pp ind^{-1}		
Fish—Round goby Neogobius melanostomus	Rhine River, border of France and Germany	Entire GI tract dissected, digested, density separation, filtered, microscopic examination, hot needle,	27	1.25 pp ind^{-1}		Roch and Brinker (2017)
Fish—common barbel *Barbus barbus*			20	1.0 pp ind^{-1}		
Fish—catfish *Hoplosternum littorale*	Pajeu River, Brazil	GI tract dissected, rinsed in 63 μm sieve, visual examination	83	3.6 pp ind^{-1} Range 1–24	Size range <1–12 mm 88.6% of items <5 mm 46.6% fibers, Remainder hard and soft particles	Silva-Cavalcanti et al. (2017)
Fish—5 species	Waucana Creek, Canada	Entire GI tract removed and digested, 5 μm filter, microscopic examination	73.5 overall	3.28 pp ind^{-1} range 1–20	48% fibers, 43% fragments	Campbell et al. (2017)
Northern Pike-*Esox lucius*			83		15% contained balls of clear fibers (not enumerated)	
White Sucker *Catostomus commersoni*			72			
Emerald Shiner *Notropis atherinoides*			71			
Fathead minnow *Pimephales promelas*			50			
Five-spine stickleback *Eucalia inconstans*			70			
Fish—13 species	Three Gorges Reservoir, China	GI tract digested, filtered, microscopic examination, Raman microscopy	25.7	0.33 ± 0.58 to 1.5 ± 1.38 pp ind^{-1} for different species	Fibers, sheets, fragments PE and nylon	Zhang et al. (2017)
Fish—roach, *Rutilus rutilus*	River Thames, UK	Entire GI tract opened, contents removed, examined microscopically, Raman	32.8	0.69 pp ind^{-1},	75% fibers PE, PP, PES	Horton et al. (2018)
Fish—11 species	Lake Michigan tributaries, USA	GI tract dissected, dried, digested, filtered, microscopic examination	85	10 ± 2.3 to 13 ±1.6 pp ind^{-1} at three sites	97–100% fibers	McNeish et al. (2018)
Fish—cyprinid *Gymnocypris przewalskii*	Qinghai Lake	GI tract dissected, digested, filtered, microscopic examination, Raman	100	5.4 pp ind^{-1} (2–15 pp ind^{-1})	Fibers in all, sheets in half, PE, PS, nylon, and PP	Xiong et al. (2018)
Fish—16 species in family Serrasalmidae (pacu and piranhas)	Xingu River Basin, Brazil	Stomachs dissected, rinsed, microscopic examination, FTIR	26.7		29.2% MP 70.8% MesoP[b] 53% filaments, 47% fragments 8 polymers	Andrade et al. (2019)

(*Continued*)

Table 4 (Continued)

Organism	*Location*	*Processing procedure*	*% Organisms with MPs*	*Average number of pp ind^{-1} or weight^{-1} of organism*	*Dominant particle morphology and polymer*	*References*
Fish—Eel *Monopterus albus*	Rice and Fish culture ponds, China	Digestive tract dissected, digested, filtered, microscopic examination, FTIR		3.3 ± 0.5 pp ind^{-1}	Fibers, PE, PP	Lv et al. (2019)
Fish—Loach *Misgurnus anguillicaudatus*	Rice and Fish culture ponds, China	Digestive tract dissected, digested, filtered, microscopic examination, FTIR		1.8 ± 0.5 pp ind^{-1}	Fibers, PE, PP	
Fish—three spine stickleback Gasterosteus aculeatus and amphibians–Newts Triturus vulgaris	Viborg stormwater retention pond, Denmark	Organisms freeze dried, pooled, digested, filtered, FTIR imaging		65 pp ind^{-1} 3 µg ind^{-1}	41% polyester, also PP, PA, PE	Olesen et al. (2019)
Fish—22 species	11 rivers and 6 lakes in Germany	GI tract dissected, digested, density separation, filtered, microscopic examination, hot needle test, only 20–5000 µm particles included	20.6% in rivers 16.5% in lakes	1–4 pp ind^{-1}	54% fragments, 39% fibers	Roch et al. (2019)
Fish—Gudgeon *Gobio gobio*	Rivers in Belgium	GI tract dissected, dried, ground, density separation, digestion, microscopic exam, FTIR or Raman	9		Only 8 particles confirmed as MP, 7 different polymers	Slootmaekers et al. (2019)
Fish—*Gambusia holbrooki*	Wetlands, Australia	Separated heads and bodies, digested separately, filtered, microscopic examination, µFTIR-ATR	19.4 in body 7.2 % in heads	0.60 ± 1.33 pp body^{-1} 0.11 ± 0.44 pp head^{-1}	62–100% fibers 11 polymers, PES, rayon, polyamide, P dominant.	Su et al. (2019)
Fish—9 species	Pearl River Catchment, China	GI tract removed, freeze-dried, digested, filtered, density separation, microscopic examination, µFTIR	50% overall	7.0 ± 23.8 pp ind^{-1}	Fibers, fragments, films common PET, PE, PP, PE/PP	Zheng et al. (2019)
Silver carp *Hypophthalmichtys molitrix*			45	2.7 ± 4.0 pp ind^{-1}		
Grass carp *Ctenopharyngodon idella*			37.5	2.8 ± 3.9 pp ind^{-1}		
Guangdong black bream *Megalobrama hoffmanni*			61.4	6.0 ± 11.3 pp ind^{-1}		
Barbel chub *Squaliobarbus curriculus*			50.0	3.0 ± 6.0 pp ind^{-1}		
Mud carp *Cirrhinus molitorella*			48.8	4.1 ± 7.5 pp ind^{-1}		
Common carp *Cyprinus carpio*			15.8	0.2 ± 0.4 pp ind^{-1}		
Prussian carp *Carrasius gibella*			43.6	1.6 ± 2.8 pp ind^{-1}		
Redbelly tilapia *Coptodon zillii*			75.0	27.4 ± 54.0 pp ind^{-1}		
Blotched snakehead Channa maculata			25.0	0.4 ± 0.8 pp ind^{-1}		
Birds—Grey heron *Ardea cinereal*, Mute swan—*Cygnus olor* Mallard *Anas platyrhynchos*	Lake Geneva, Switzerland	Digestive tract removed, examined	89	4.3 ± 2.6 pp ind^{-1} 4.8 ± 8.9 mg ind^{-1}	Fragments dominant, foam, film, beads, fiber also observed	Faure et al. (2015)

[a]*pp*: plastic particle; *ind*: individual; *dw:* dry weight; *ww*: wet weight.
[b]Mesoplastics ranged from 5.1 to 25 mm in length.
Data sources: See References.

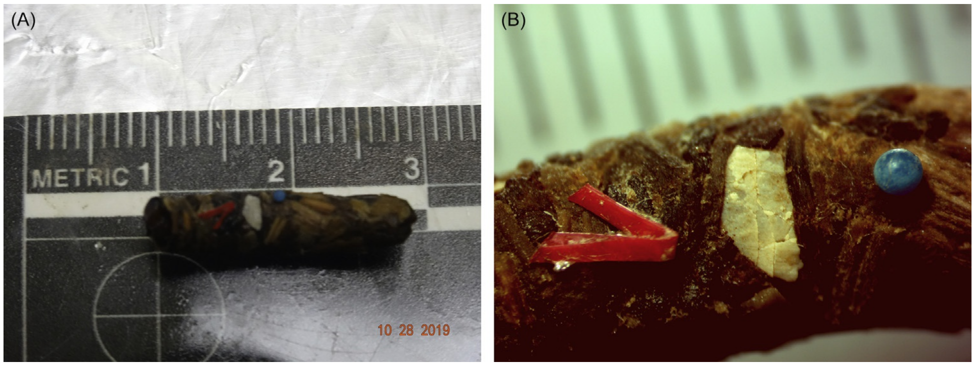

Fig. 4 Limnephilidae caddisfly case with MPs added by authors to show sizes of natural debris with plastic particles.

also reported that the use of fluorescence alone is not sufficient to prove that MP particles have moved from the gut into other organs, especially for particles >1 μm in size.

MPs can affect aquatic organisms in a variety of ways. Canniff and Hoang (2018) found that algal growth increased in the presence of PE beads which may have served as a substrate for the algae. On the other hand, Lagarde et al. (2016) reported that algal growth was reduced after 78 days of exposure to PP and HDPE fragments, hypothesizing that algal colonization and aggregation of the MPs increased the density of the fragments which settled out of the water column.

Many scientists have employed standardized toxicity test methods to examine the effects of MP on amphipods, water fleas, and fish. These experiments exposed test organisms to a range of MP concentrations under controlled conditions and then determined the highest concentration that had no effect of the organisms (NOEC), the lowest concentration that initiated an observed effect (LOEC), and the concentrations that caused an effect or were lethal to 50% of the test organisms (EC50 and LC50 respectively). Effects can include reduced feeding rates, motility, growth, reproduction, or survival.

Redondo Hasselerharm et al. (2018) reported that only one of six benthic invertebrate species they tested were affected by MPs when exposed to PS fragments in sediments at concentrations up to 40% of the sediment dry weight (Table 5). The amphipod *Hyalella azteca* was more sensitive to PP fibers (10-day LC50 = 71 pp mL^{-1}) than to PE particles (10-day LC50 = 46,400 pp mL^{-1}, Au et al., 2015). Ziajahromi et al. (2018) found that growth and survival of the midge *Chironomus* was affected by MP at environmentally realistic concentrations (0.5 pp g^{-1}) in sediments.

MP can interfere with some organisms' ability to swim or feed by becoming entangled in their appendages or filling their guts (Table 5). Eltemsah and Bøhn (2019) reported a 5-day EC50 mobility of 35–52 mg L^{-1}, for the waterflea, *Daphnia*, exposed to 6 μm PE beads. PE fragments affected *Daphni*a reproduction more than PE beads (21-day EC50 = 8.6×10^4 pp mL^{-1} versus 2.8×10^5 pp mL^{-1}, Ogonowski et al., 2016). *Ceriodaphnia* are more sensitive than *Daphnia* to MP pollution (Ziajahromi et al., 2017a) and showed greatest sensitivity to PES fibers than to other forms. Although the fibers were not ingested, they interfered with *Ceriodaphnia* swimming and led to a 48-h LC50 of 1.5 mg L^{-1} or 1.3×10^4 pp L^{-1}. In chronic 21-day tests, Jaikumar et al. (2019) found that MP concentrations as low as 100 pp mL^{-1} decreased at least one component of reproduction in three species of water fleas.

More sensitive biological tests have been used to detect evidence of physiological effects (Table 5) including oxidative stress, neurotoxicity, and immune responses in model organisms including fish (Karami et al., 2016; Lu et al., 2016; Haghi and Banaee, 2017; Jin et al., 2018; Lei et al., 2018; Wen et al., 2017).

Adverse effects in the environment can be caused by exposure of organisms to MP particles themselves or to pollutants that have been absorbed by the MPs. Laboratory studies have been conducted to examine the combined effects of MPs and a variety of chemical contaminants (Table 6). Wardrop et al. (2016) and (Rochman et al., 2013b) fed fish a diet that contained MP particles that had been exposed to PBDE, PAHs, and PCBs in the environment. Bioaccumulation of the contaminants in the test organisms was enhanced for some of the congeners. In experiments where organisms were exposed to MP and chemical contaminants in the same test solution, the MPs often increased the uptake and toxicity of the chemicals (Guilhermino et al., 2018; Frydkjær et al., 2017; Karami et al., 2016; Haghi and Banaee, 2017) or the effects were additive (Ma et al., 2016).

Ziajahromi (2018) found that MPs may decrease the toxicity of insecticides to midge larvae in lab water, but did not see the same effect in river water where higher organic carbon levels may have had a greater affinity for the insecticide. Understanding the chemical partitioning rates between chemical contaminants and plastics versus natural organic carbon is critical for predicting the effects of MP on chemical transfer in aquatic ecosystems. Frydkjær et al. (2017) found that PE fragments absorbed phenanthrene at a rate 8–35 times lower than natural plankton under laboratory conditions and they concluded that since the abundance of plastic in aquatic environments is much less than that of plankton, plastic may not be significant route of enhanced chemical exposure to organisms in the food chain.

Table 5 Laboratory experiments on effects of MPs on freshwater organisms.

Organism	*Exposure conditions*		*Results*	*References*
	Particle type[a] *and size (μm)*	*Plastic concentration*		
Bacteria				
Microbial community	PE spheres 212–250	2 and 20 mg g^{-1} sediment ≈ 257 and 2570 pp g^{-1} sediment	Reduced diversity of microbial community in clean sediment in 14 days	Kleinteich et al. (2018)
Algae				
Chlamydomonas reinhardtii	PP and HDPE fragments 400–1000	100 mg in 100 mL media	NEF on growth in 60 days, aggregates of algae and pp developed, increased settling rate of algae, may account for reduced growth in 78 days	Lagarde et al. (2016)
Raphidocellis subcapitata	PE beads 63–75	130 mg L^{-1} ≈ 9.9×10^6 pp L^{-1}	Increased algal growth, possibly served as substrate	Canniff and Hoang (2018)
Plant				
Lemna minor	PE spheres 10–45	5.0×10^4 pp mL^{-1}	MP adhered to *Lemna* surfaces-maximum 7 pp mm^{-2} or 42 pp dried $colony^{-1}$. NEF photosynthesis or growth in 30 days	Mateos-Cárdenas et al. (2019)
Cnidaria				
Hydra attenuata	PE flakes <400	0.01–0.08 g mL^{-1} ~800–6400 pp mL^{-1}	Particles were ingested, reduced feeding rates, increased egestion rates Predicted NOEC 2800 pp mL^{-1}	Murphy and Quinn (2018)
Mollusca				
Asian clam *Corbicula fluminea*	PET fragments 12–704	4.1 mg L^{-1}	Accumulated 5 ± 6 pp ind^{-1} in 3 days	Rochman et al. (2017)
	PE fragments 14–704	2.8 mg L^{-1}	Accumulated 8 ± 6 pp ind^{-1} in 3 days	
	PVC fragments 80–704	4.2 mg L^{-1}	Accumulated 3 ± 3 pp ind^{-1} in 3 days	
	PS fragments 68–704	3.2 mg L^{-1}	Accumulated 4 ± 3 pp ind^{-1} in 3 days	
	Fluorescent polymer spheres 1–5	0.2 and 0.7 mg L^{-1} ≈ 3.67×10^7–1.28×10^8 pp L^{-1}	MP found in digestive tract, digestive gland, connective tissues, hemolymphatic sinuses and on gills. Inhibition of enzyme activity	Guilhermino et al. (2018)
	PEA fibers	100 and 1000 pp L^{-1}	0.3 pp g^{-1} ingested at 100 and 1000 pp L^{-1} Fibers arched	Li et al. (2019)
	AC fibers	100 and 1000 pp L^{-1}	Ingested at 1000 pp L^{-1}	
	PA fibers	100 and 1000 pp L^{-1}	0.1 pp g^{-1} ingested at 1000 pp L^{-1}	
	RA fibers	100 and 1000 pp L^{-1}	Not ingested	
	PVA fibers	100 and 1000 pp L^{-1}	Not ingested	
	PET fibers	100 and 1000 pp L^{-1}	Ingested 0.5 pp g^{-1} at 100 pp L^{-1} and 4.1 pp g^{-1} at 1000 pp L^{-1} Ingested fibers were curved and bent into loops	
	PET fibers 100–250	100 and 1000 pp L^{-1}	Ingested 1.7 pp g^{-1} at 1000 pp L^{-1}	
	PET fibers 1000–5000	100 and 1000 pp L^{-1}	Ingested 0.2 pp g^{-1} at 1000 pp L^{-1}	
Zebra mussel *Dreissena polymorpha*	PS beads 1 and 10	5×10^5 pp L^{-1} of each size and 2×10^6 pp L^{-1} of each size	MP found in gut lumen, tissues, and hemolymph. Increased level of dopamine, did not produce oxidative stress or genetic damage.	Magni et al. (2018)
Mud snail *Potamopyrgus antipodarum*	Polymethyl methacrylate fragments 29.5	1:10 ratio of plastic to food	MP ingested, 87.8% of feces contained MPs	Imhof et al. (2013)
European fingernail clam *Sphaerium corneum*	PS fragments 20–500	0.1–40% of sediment dw	NEF survival or growth in 28 days	Redondo Hasselerharm et al. (2018)

Annelida				
Lumbriculus variegatus	Polymethyl methacrylate fragments 29.5	1:10 ratio of plastic to food	93% ingested MPs	Imhof et al. (2013)
	PS fragments 20–500	0.1–40% of sediment dw	NEF growth or reproduction in 28 days	Redondo Hasselerharm et al. (2018)
Tubifex spp.	PS fragments 20–500	0.1–40% of sediment dw	NEF survival or growth in 28 days	Redondo Hasselerharm et al. (2018)
Nematoda				
Caenorhabditis elegans	PA fragments 70	0.5–10.0 mg m^{-2}	Decreased survival, growth, reproduction, and intestinal calcium levels. Oxidative stress noted. LOEC $= 0.5$ mg m^{-2}	Lei et al. (2018)
	PE fragments 70	0.5–10.0 mg m^{-2}	Decreased survival, growth, reproduction, and intestinal calcium levels. Oxidative stress noted. LOEC $= 0.5$ mg m^{-2}	
	PP fragments 70	0.5–10.0 mg m^{-2}	Decreased survival, growth, reproduction, and intestinal calcium levels. Oxidative stress noted. LOEC $= 0.5$ mg m^{-2}	
	PVC fragments 70	0.5–10.0 mg m^{-2}	Decreased survival, growth, reproduction, and intestinal calcium levels. Oxidative stress noted. LOEC $= 1.0$ mg m^{-2}	
	PS spheres 0.1, 1.0, 5.0	0.5–10.0 mg m^{-2}	PP accumulated in digestive system. Caused oxidative stress, decreased survival, growth, reproduction, and intestinal calcium levels. 1.0 μm particles most lethal. LOEC $= 0.5$ mg m^{-2}	
Crustaceans				
Ostracod *Notodromas monacha*	Polymethyl methacrylate fragments 29.5	1:1 ratio of plastic to food	32.4% ingested MPs	Imhof et al. (2013)
Isopod *Asellus aquaticus*	PS fragments 20–500	0.1–40% of sediment dw	NEF survival or growth in 28 days	Redondo Hasselerharm et al. (2018)
Amphipod *Gammarus duebeni*	PE spheres 10–45	Fed *Lemna minor* with adsorbed MP	29% contained pp after 24 h depuration period, NEF survival in 48 h exposure	Mateos-Cárdenas et al. (2019)
Amphipod *Gammarus fossarum*	PA fibers 500 × 20	100–1.3 × 10^4 fibers cm^{-2} basal area of chamber	Ingested and egested rapidly. NEF feeding rate. Decreased assimilation efficiency at 2680 fibers cm^{-2}	Blarer and Burkhardt-Holm (2016)
	PS beads 1.6	500–6 × 10^4 pp mL^{-1}	PP ingested, found in gut but not in midgut glands or epithelial cells of gut. NEF feeding rate or assimilation efficiency at 12,500 beads mL^{-1}	
	PHB fragments (biodegradable) 32–250	10–1 × 10^5 pp ind^{-1}	32–63 μm particles ingested in highest amounts, NEF feeding rate or assimilation efficiency. Decreased weight gain	Straub et al. (2017)
	PMMA fragments 32–250	10–1 × 10^5 pp ind^{-1}	32–63 μm particles ingested in highest amounts, decreased assimilation efficiency and weight gain	
Amphipod *Gammarus pulex*	Polymethyl methacrylate 29.5	1:10 ratio of plastic to food	Ingested, 96% of feces contained MPs	Imhof et al. (2013)
	PS fragments 20–500	0.1–40% of sediment dw	16–165 μm MPs were ingested. Ingestion rate proportional to sediment dose. NEF survival or feeding rate in 28 days. EC10 growth $= 1.07\%$ EC50 growth $= 3.57\%$	Redondo Hasselerharm et al. (2018)

(*Continued*)

Table 5 (Continued)

Organism	Exposure conditions		Results	References
	Particle type and size (μm)	Plastic concentration		
	PET fragments 10–150	0.4–4000 pp mL^{-1}	High ingestion of fragments <53 μm, up to 6600 pp ind^{-1} NEF survival, metabolism or feeding in 48 days	Weber et al. (2018)
Amphipod *Hyalella azteca*	PE particles 10–27	10 - 1 $\times$ 10^5 pp mL^{-1} acute exposure 5 $\times$ 10^3 – 2 $\times$ 10^4 pp mL^{-1} chronic exposure	MP ingested, but did not translocate out of gut. NEF growth in acute exposure. Decreased reproduction and growth in chronic exposure. 10-day LC50 = 46,400 pp mL^{-1}	Au et al. (2015)
	PP fibers diameter 20 length 20–75	22.5–90 pp mL^{-1}	MP ingested, but did not translocate out of gut. Increased egestion time. Decreased growth. 10-day LC50 = 71.43 pp mL^{-1}	
	PS fragments 20–500	0.1–40% of sediment dw	No MP ingestion. NEF survival, feeding rate or growth in 28 days.	Redondo Hasselerharm et al. (2018)
Water flea *Ceriodaphnia dubia*	PE beads 1–4	0.5–16 mg L^{-1} $\approx$1.7 $\times$ 10^4–5.4 $\times$ 10^5 pp L^{-1}	Beads ingested, gut full at 4 mg L^{-1} and higher 48-h LC50 = 2.2 mg L^{-1} (7.4 $\times$ 10^4 pp L^{-1})	Ziajahromi et al. (2017a)
	PE beads 1–4	62.5–2000 μg L^{-1} $\approx$2.1 $\times$ 10^3–6.7 $\times$ 10^4 pp L^{-1}	8-day EC50 reproduction = 958 μg L^{-1} (3.2 $\times$ 10^3 pp L^{-1})	
	PES fibers 26–1150 long average 280	0.125–4 mg L^{-1} $\approx$1.1 $\times$ 10^3–3.4 $\times$ 10^4 pp L^{-1}	No fibers ingested, but bubbles noted under carapace, antennal and carapace deformities, fibers affected swimming behavior 48-h LC50 = 1.5 mg L^{-1} (1.3 $\times$ 10^3 pp L^{-1})	
	PES fibers 26–1150 long average 280	31.25–1000 μg L^{-1} $\approx$272–8600 pp L^{-1}	8-day EC50 reproduction = 429 μg L^{-1} (3.5 $\times$ 10^3 pp L)	
	PE spheres 1–5	1 $\times$ 10^3–1 $\times$ 10^7 pp mL^{-1}	18 °C 96-h NOEC= 5.0 $\times$ 10^3 pp mL^{-1} Little change at higher temperatures	Jaikumar et al. (2018)
	PE fragments 1–10	1 $\times$ 10^3–1 $\times$ 10^7 pp mL^{-1}	18 °C 96-h NOEC = 1.0 $\times$ 10^5 pp mL^{-1} Little change at higher temperatures	
	PE spheres 1–5	100–1 $\times$ 10^5 pp mL^{-1}	21-day LOEC reproduction = 100 pp mL^{-1}	Jaikumar et al. (2019)
	PE fragments 1–10	100–1 $\times$ 10^5 pp mL^{-1}	21-day LOEC reproduction = 100 pp mL^{-1}	
Water flea *Daphnia magna*	Fluorescent PS beads 0.02 and 1.00	2 μg L^{-1}	Both sizes ingested. Fluorescence appeared in gut and lipid storage droplets. Depuration rate faster for the larger MP beads	Rosenkranz et al. (2009)
	PMA fragments 29.5	1:10 ratio of plastic to food	100% ingested MP	Imhof et al. (2013)
	PET fibers 60–1400 long, 30–530 wide, 2–21.5 thick	12.5–100 mg L^{-1} $\approx$6.64 $\times$ 10^4–5.3 $\times$ 10^5 fibers L^{-1}	MP uptake and depuration noted, increased mortality in 48 h in unfed organisms- not dose dependent.	Jemec et al. (2016)
	PS 0.05–10	1–50 mg L^{-1}	Uptake and depuration of MP and NP observed. NEF from larger particles. 0.05 μm NP most toxic. 48-h EC50 immobilization = 15.13 mg L^{-1}, damaged swimming/feeding appendages.	Ma et al. (2016)
	PE beads 4.1 $\pm$ 1.0	100–2.25 $\times$ 10^5 pp mL^{-1}	Decreased feeding rate, and growth, increased mortality. 21-day EC50 reproduction = 2.8 $\times$ 10^5 pp mL^{-1}	Ogonowski et al. (2016)
	PE fragments 2.6 $\pm$1.8	100–2.25 $\times$ 10^5 pp mL^{-1}	Decreased feeding rate, and growth, increased mortality. 21-day EC50 reproduction = 8.6 $\times$ 10^4 pp mL^{-1}	
	PE spheres 1–4	12.5–400 mg L^{-1}	Ingestion and egestion noted, caused immobilization 96-h EC50 = 57.43 mg L^{-1}	Rehse et al. (2016)
	PE spheres 90–106	12.5–400 mg L^{-1}	MP accumulated on water surface, not ingested, NEF	

	PS beads 2	0.1–1 mg L^{-1} ≈1.4 × 10^6–1.4 × 10^7 pp L^{-1}	Ingested up to 1.24 × 10^6 pp ind^{-1} or 0.89 μg ind^{-1} in 24 h. In 21 days, body burden up to 3.0 × 10^5 pp ind^{-1} or 2.43 μg ind^{-1} NEF survival or reproduction	Rist et al. (2017)
	PE beads 10–106 and PE fragments 10–75	0.0001–10 g L^{-1}	24 h ingestion related to dose 33–50 pp ind^{-1} in 24 h, Beads egested at higher rate than fibers 48-h EC50 mobility =0.065 g L^{-1} for fragments	Frydkjær et al. (2017)
	PE beads 63–75	25–100 mg L^{-1} ≈1.9 × 10^6–7.6 × 10^6 pp L^{-1}	Ingested 3.81–15.06 pp ind^{-1} NEF survival or reproduction	Canniff and Hoang (2018)
	PE spheres 1–5	1 × 10^3–1 × 10^7 pp mL^{-1}	18 °C 96-h NEC=1 × 10^5 pp mL^{-1} More toxic at higher temperatures	Jaikumar et al. (2018)
	PE fragments 1–10	1 × 10^3–1 × 10^7 pp mL^{-1}	18 °C 96-h NEC= 5.01 × 10^4 pp mL^{-1} More toxic at higher temperatures	
	PS beads 6	5–300 mg L^{-1}	MP ingested and adhered to body surfaces and appendages. Decreased growth. 120-h EC50 immobilization = 35–52 mg L^{-1}	Eltemsah and Bøhn (2019)
	PE spheres 1–5	100–1 × 10^5 pp mL^{-1}	21-day LOEC reproduction = 100 pp mL^{-1}	Jaikumar et al. (2019)
	PE fragments 1–10	100–1 × 10^5 pp mL^{-1}	21-day LOEC reproduction = 100 pp mL^{-1}	
	PS beads 0.02 and 1.0	2 μg L^{-1}	Did not detect smaller fluorescent particles in Daphnia. Larger particles clearly visible in digestive tract. No fluorescence outside of gut.	Schür et al. (2019)
		2 mg L^{-1}	Fluorescence observed in lipid droplets and gut of Daphnia for both sizes of MP. Larger beads observed in gut. Fluorescence of lipids due to movement of dye, not pp.	
	PS 1	0.1–600 mg L^{-1}	48-h EC50 immobilization = 66.97 mg L^{-1} 48-h LC50 = 87.83 mg L^{-1}	Zhang et al. (2019a)
	PS 10	0.005–40 mg L^{-1}	48-h EC50 immobilization = 19–9.94 mg L^{-1} 48-h LC50 = 291.69 mg L^{-1}	
	PET/PA fibers 10 × 2	10 pp mL^{-1}	31.7% mortality in 168 h Aggregates of MP and algae noted on antennae and carapace	Zocchi and Sommaruga (2019)
	PE beads 1–10	0.01 mg mL^{-1} ≈ 2.2 × 10^6 pp mL^{-1}	38% mortality in 168 h Aggregates of MP and algae noted on antennae and carapace	
Daphnia pulex	PE spheres 1–5	1 × 10^3–1 × 10^7 pp mL^{-1}	18 °C 96-h NOEC = 1 × 10^5 pp mL^{-1} More toxic at higher temperatures	Jaikumar et al. (2018)
	PE fragments 1–10	1 × 10^3–1 × 10^7 pp mL^{-1}	18 °C 96-h NOEC= 1 × 10^5 pp mL^{-1} More toxic at higher temperatures	
	PE spheres 1–5	100–1 × 10^5 pp mL^{-1}	21-day LOEC reproduction = 100 pp mL^{-1}	Jaikumar et al. (2019)
	PE Fragments 1–10	100–1 × 10^5 pp mL^{-1}	21-day LOEC reproduction = 100 pp mL^{-1}	
Insects				
Mosquito *Culex pipiens*	PS beads 2 and 15	50–200 pp L^{-1}	Larvae ingested MP—up to 256 pp $larva^{-1}$. MP transferred to pupal and adult stages. NEF survival or growth	Al-Jaibachi et al. (2019)
Mosquito *Culex pipiens*	PS 2	100 pp mL^{-1}	MP ingested, average 5.8 pp ind^{-1}	Cuthbert et al. (2019)
Phantom midge *Chaoborus flavicans*	PS 2	Fed *C. pipiens* that had been exposed to MP	MP found in predators, amount related to consumption rate	
Midge *Chironomus tepperi*	PE spheres 1–4, 10–27, 43–54, 100–126	500 pp kg^{-1} sediment ww	NEF on survival or growth with 100–126 μm MP. Decreased survival and growth in three smaller size classes, effects greatest for 10–27 μm MP. Emergence of adults decreased for all MP sizes	Ziajahromi et al. (2018)
Fish				
African catfish *Clarias gariepinus*	LDPE fragments < 60	50 and 500 μg L^{-1} ≈1.4 × 10^3 and 1.4 × 10^4	MP not found in gill or liver tissues but gill and liver tissues were damaged. Altered levels of lipids and proteins in blood plasma Decreased gene transcription in brain tissue	Karami et al. (2016)
Common carp *Cyprinus carpio*	PE	1–2 mg L^{-1}	Changes in blood biochemistry indicating physiological stress and organ damage	Haghi and Banaee (2017)

(*Continued*)

Table 5 (Continued)

Organism	Exposure conditions		Results	References
	Particle type and size (μm)	*Plastic concentration*		
Zebrafish *Danio rerio*	PS beads 5	20 mg L^{-1} 2.9×10^5pp mL^{-1}	MP accumulated in gills, liver, and gut	Lu et al. (2016)
	PS beads 20	20 mg L^{-1} 4500 pp mL^{-1}	MP accumulated in gills and gut	
	PS beads 0.07, 5	20–2000 μg mL^{-1}	High doses caused inflammation and signs of oxidative stress in the liver. Lipid profile and liver metabolites affected	
	PS beads 0.5 and 50	100 and 1000 μg L^{-1}	At higher concentrations MP caused inflammation and increased production of mucous in gut and resulted in changes in gut microbiota	Jin et al. (2018)
	PA fragments 70	0.001–10.0 mg L^{-1}	NEF survival, caused intestinal damage	Lei et al. (2018)
	PE fragments 70	0.001–10.0 mg L^{-1}	NEF survival, caused intestinal damage	
	PP fragments 70	0.001–10.0 mg L^{-1}	27% decrease survival at highest concentration, caused intestinal damage	
	PVC fragments 70	0.001–10.0 mg L^{-1}	NEF survival, caused intestinal damage	
	PS spheres 0.1, 1.0, 5.0	0.001–10.0 mg L^{-1}	NEF survival or intestine	
Discus fish *Symphysodon aequifasciatus*	PE spheres 70–88	200 μg L^{-1}	NOEF on growth or survival in 30 days at temperatures of 28 °C or 31 °C. Greater accumulation of MP in tissue at higher temperature. MP affected predation rates at lower temp. Digestive and metabolic enzyme activities altered by presence of MP	Wen et al. (2017)

[a]Polymer types: *PE*: polyethylene; *PEA*: polyester amide; *PES*: polyester; *PHB*: polyhydroxybutyrate (biodegradeable bioMP); *PMA*: Polymethyl acrylate; *PVA*: polyvinyl alcohol; *RA*: rayon.
Data sources: See References.

Table 6 Interactions of MP with chemical contaminants in laboratory exposures.

Species	MP exposure: Particle type[a] and size μm	MP exposure: Plastic concentration	Chemical contaminant[b] and exposure concentration	Results	Reference
Microbial community	PE spheres 212–250	1 mg g^{-1} sediment 257 pp g^{-1} sediment	Phenanthrene 30.2 μg g^{-1} on MP	Observed change in sediment microbial community structure exposed to contaminants on MPs was different than response to contaminant alone	Kleinteich et al. (2018)
			Anthracene 29.5 μg g^{-1} on MP	No change in microbial community structure, but MP decreased rate of anthracene degradation.	
Asian clam *Corbicula fluminea*	Fluorescent polymer spheres 1–5	0.2 and 0.7 mg L^{-1} 3.67×10^7 to 1.28×10^8 pp L^{-1}	Florfenicol 1.8 and 7.1 mg L^{-1} in test solution	MP increased uptake of florfenicol. Decreased feeding and cholinesterase activity, showed neurotoxicity, oxidative damage	Guilhermino et al. (2018)
Water flea *Daphnia magna*	PS 0.05–10	2.5–50 mg L^{-1}	Phenanthrene 0.05–1.2 mg L^{-1} in test solution	Joint effects of MP and phenanthrene were additive. MP increased immobilization of *Daphnia.* Increased bioaccumulation of phenanthrene, and slowed degradation rate. MP did not bioaccumulate or affect toxicity of phenanthrene.	Ma et al. (2016)
	PE fragments 10–75	0.05 g L^{-1}	Phenanthrene 0.008–5 mg L^{-1} in test solution	MP increased toxicity Phenanthrene EC50 = 0.47 mg L^{-1} w/o MP and 0.14 mg L^{-1} w/MP	Frydkjær et al. (2017)
	PA fragments 15–20	200 mg L^{-1}	BPA 5–15 mg L^{-1} on MP	MPs adsorbed BPA and were ingested, MPs reduced rate of immobilization from BPA.	Rehse et al. (2018)
	PS 1	0.1 mg L^{-1} 1.82×10^8 pp mL^{-1}	Roxithromycin 0.01 mg L^{-1} in test solution	MP increased oxidative stress, decreased antioxidant enzyme activity	Zhang et al. (2019a)
	PS 10	0.1 mg L^{-1} 1.82×10^5 pp mL^{-1}		MP reduced oxidative stress, Increased antioxidant enzyme activity	
	PET/PA fibers 10 × 2	10 pp mL^{-1}	Glyphosate acid 2.5 mg L^{-1} in test solution	Increased Mortality in 168 h	Zocchi and Sommaruga (2019)
			Roundup gran 2.5 mg L^{-1} in test solution	Increased mortality in 168 h	
			Glyphosate-IPA 2.5 mg L^{-1} in test solution	Decreased mortality in 168 h	
	PE beads 1–10	2,200,000 pp mL^{-1}	Glyphosate acid 2.5 mg L^{-1} in test solution	Increased mortality in 168 h	
		0.01 mg mL^{-1}	Roundup gran 2.5 mg L^{-1} in test solution	Increased mortality in 168 h	
			Glyphosate-IPA 2.5 mg L^{-1} in test solution	Decreased mortality in 168 h	
Midge *Chironomus tepperi*	PE beads 10–27	1600 pp L^{-1} 5 mg L^{-1}	Bifenthrin 0.1–3.2 μg L^{-1} in test solution	Decreased toxicity of bifenthrin in lab media, but not in river water. 48-h LC50 =1.3 μg L^{-1} with MP vs. 0.5 μg L^{-1} w/o MP	Ziajahromi (2018)
Zebrafish *Danio rerio*	PE 10–106	10–1000 pp mL^{-1}	Silver 1 μg L^{-1} in test solution	MP did not affect uptake of Ag in 24 h exposure	Khan et al. (2015)
	PE 10–106	1000 pp mL^{-1}	Silver Beads incubated with l μg L^{-1} Ag prior to exposure to 1 μg L^{-1} in test solution	MP reduced total uptake of Ag in 24 h, but increased proportion in intestine	
	PVC 200–250	400 mg L^{-1}	Phenanthrene 0.5 mg L^{-1} in test solution	MP addition reduced gene expression indicating reduced phenanthrene availability	Sleight et al. (2017)
			17α-ethinylestradiol 1 μg L^{-1}in test solution	MP addition reduced gene expression indicating reduced ethinylestradiol availability	

(*Continued*)

Table 6 (Continued)

Species	*MP exposure*		*Chemical contaminant and exposure concentration*	*Results*	*Reference*
	Particle type and size μm	*Plastic concentration*			
Japanese medaka *Oryzias latipes*	PE Virgin and exposed/ aged pellets <500	10% by weight of diet Fed 6 mg food with 0.6 mg MP fish^{-1} day^{-1}	PAHs, 24–58 ng g^{-1} diet PCBs 0.3–5.3 ng g^{-1} diet PBDEs 1.1–3.1 ng g^{-1} diet	Body burden in exposures with contaminated MP similar to controls except for elevated chrysene, PCB28, and PBDE levels. Liver damage noted indicating glycogen depletion	Rochman et al. (2013b)
African catfish *Clarias gariepinus*	LDPE fragments <60	50 and 500 μg L^{-1} (1400 and 14,000 PP L^{-1})	Phenanthrene Nominal concentrations 10 and 100 μg L^{-1} on MP	MP may have changed bioavailability of phenanthrene. Increased degree of gill damage, changes in lipids and proteins in blood plasma and gene transcription rates in brain compared to phenanthrene alone.	Karami et al. (2016)
Common carp *Cyprinus carpio*	PE	1–2 mg L^{-1}	Paraquat 0.2–0.4 mg L^{-1} in test solution	Changes in blood biochemistry indicating physiological stress and organ damage. Increased toxicity of Paraquat	Haghi and Banaee (2017)
Rainbow fish *Melanotaenia fluviatilis*	PE 10–700	10 mg day^{-1} in food	PBDE 200 ng g^{-1} day^{-1} diet on MP	MP ingested, PBDE bioaccumulated in fish tissues over 63 days, amount varied by PBDE congener	Wardrop (2016)

[a]*PE*, polyethylene
[b]*BPA*, bisphenol A; *PAH*, polycyclic aromatic hydrocarbon; *PBDE*, polybrominated diphenyl ethers; *PCB*, polychlorinated biphenyl.
Data sources: See References.

Summary

The increased use of plastic materials by humans has led to the contamination of freshwater biomes with MP debris. MP particles are now found in the water column and sediments of lakes and rivers worldwide. While concentrations of MPs are generally fairly low <1 pp L^{-1} in surface waters and <10 pp g^{-1} sediment, higher concentrations are observed near urban areas, sewage plants and retention ponds where materials can settle. With the use of finer nets and sieves for sample collection, researchers are finding that fibers are more widespread and numerous than fragments or microbeads and pellets.

Although freshwater organisms do ingest MPs, the density of plastic in individual organisms in nature is generally quite low due to the ability of the organisms to depurate the plastic materials. Laboratory studies have found that MPs can adversely affect the metabolism, growth, reproduction, and survival of aquatic organisms, but generally at concentrations much higher than those found in nature.

Few studies have been done on the ability of MPs to accumulate toxic chemicals from freshwater environments and the kinetics of the release of these compounds to organisms upon ingestion is generally unknown under environmental conditions. Koelmans et al. (2016) found that MPs adsorb POPs in lower concentrations that natural media in the oceans. However, it should be noted that MPs are not biodegradable in a reasonable human time scale, and will be continuously concentrating toxic compounds from the aquatic environment.

Because MP pollution is an anthropogenic issue, we need to develop a way to decrease this environmental problem by enacting one of the UN's sustainable development goals "responsible consumption and production" (United Nations, 2015). By promoting education to raise the awareness of the general public to the negative effects of the indiscriminate use of everyday plastic items, we can produce a change in behavior regarding the consumption and uses of plastics.

References

Alam FC, Sembiring E, Muntalif BS, and Suendo V (2019) Microplastic distribution in surface water and sediment river around slum and industrial area (case study: Ciwalengke River, Majalaya district, Indonesia). *Chemosphere* 224: 637–645.
Alimi OS, Farner Budarz J, Hernandez LM, and Tufenkji N (2018) Microplastics and nanoplastics in aquatic environments: Aggregation, deposition, and enhanced contaminant transport. *Environmental Science & Technology* 52(4): 1704–1724.
Al-Jaibachi R, Cuthbert RN, and Callaghan A (2019) Examining effects of ontogenic microplastic transference on Culex mosquito mortality and adult weight. *Science of the Total Environment* 651: 871–876.
Anderson JC, Park BJ, and Palace VP (2016) Microplastics in aquatic environments: Implications for Canadian ecosystems. *Environmental Pollution* 218: 269–280.
Andrade MC, Winemiller KO, Barbosa PS, et al. (2019) First account of plastic pollution impacting freshwater fishes in the Amazon: Ingestion of plastic debris by piranhas and other serrasalmids with diverse feeding habits. *Environmental Pollution* 244: 766–773.
Andrady AL (2011) Microplastics in the marine environment. *Marine Pollution Bulletin* 62: 1596–1605.

Arias-Andres M, Klümper U, Rojas-Jimenez K, and Grossart HP (2018) Microplastic pollution increases gene exchange in aquatic ecosystems. *Environmental Pollution* 237: 253–261.
Arthur C, Baker J, and Bamford H (eds.) (2009) *Proceedings of the International Research Workshop on the Occurrence, Effects, and Fate of Microplastic Marine Debris, Sept. 9–11 2008*, . NOAA Technical Memorandum NOS-OR&R-30.
Au SY, Bruce TF, Bridges WC, and Klaine SJ (2015) Responses of *Hyalella azteca* to acute and chronic microplastic exposures. *Environmental Toxicology and Chemistry* 34: 2564–2572.
Baldwin AK, Corsi SR, and Mason SA (2016) Plastic debris in 29 Great Lakes tributaries: Relations to watershed attributes and hydrology. *Environmental Science & Technology* 50: 10377.
Ballent A, Corcoran PL, Madden O, Helm PA, and Longstaffe FJ (2016) Sources and sinks of microplastics in Canadian Lake Ontario nearshore, tributary and beach sediments. *Marine Pollution Bulletin* 110: 383–395.
Barnes DKA, Galgani F, Thompson RC, and Morton B (2009) Accumulation and fragmentation of plastic debris in global environments. *Philosophical Transactions of the Royal Society B: Biological Science* 364: 1985–1998.
Barrows AP, Christiansen KS, Bode ET, and Hoellein TJ (2018) A watershed-scale, citizen science approach to quantifying microplastic concentration in a mixed land-use river. *Water Research* 147: 382–392.
Battulga B, Kawahigashi M, and Oyuntsetseg B (2019) Distribution and composition of plastic debris along the river shore in the Selenga River basin in Mongolia. *Environmental Science and Pollution Research International* 26: 14059–14072.
Biginagwa FJ, Mayoma BS, Shashoua Y, Syberg K, and Khan FR (2016) First evidence of microplastics in the African Great Lakes: Recovery from Lake Victoria Nile perch and Nile tilapia. *Journal of Great Lakes Research* 42: 146–149.
Blair R, Waldron S, Phoenix V, and Gauchotte-Lindsay C (2019) Microscopy and elemental analysis characterisation of microplastics in sediment of a freshwater urban river in Scotland, UK. *Environmental Science and Pollution Research* 26: 12491–12504.
Blarer P and Burkhardt-Holm P (2016) Microplastics affect assimilation efficiency in the freshwater amphipod *Gammarus fossarum*. *Environmental Science and Pollution Research* 23: 23522–23532.
Blettler M, Ulla M, Rabuffetti A, and Garello N (2017) Plastic pollution in freshwater ecosystems: Macro-, meso-, and microplastic debris in a floodplain lake. *Environmental Monitoring and Assessment* 189: 1–13.
Bordós G, Urbányi B, Micsinai A, et al. (2019) Identification of microplastics in fish ponds and natural freshwater environments of the Carpathian basin, Europe. *Chemosphere* 216: 110–116.
Browne MA, Crump P, Niven SJ, et al. (2011) Accumulation of microplastic on shorelines worldwide: Sources and sinks. *Environmental Science and Technology* 45: 9175–9179.
Campbell SH, Williamson PR, and Hall BD (2017) Microplastics in the gastrointestinal tracts of fish and the water from an urban prairie creek. *FACETS* 2: 395–409.
Canniff PM and Hoang TC (2018) Microplastic ingestion by *Daphnia magna* and its enhancement on algal growth. *Science of the Total Environment* 633: 500–507.
Carr SA, Liu J, and Tesoro AG (2016) Transport and fate of microplastic particles in wastewater treatment plants. *Water Research* 91: 174–182.
Castaneda RA, Avlijas S, Simard MA, and Ricciardi A (2014) Microplastic pollution in St. Lawrence River sediments. *Canadian Journal of Fisheries and Aquatic Sciences* 71: 1767–1771.
Catarino AI, Frutos A, and Henry TB (2019) Use of fluorescent-labelled nanoplastics (NPs) to demonstrate NP absorption is inconclusive without adequate controls. *Science of the Total Environment* 670: 915–920.
Corcoran PL, Norris T, Ceccanese T, et al. (2015) Hidden plastics of Lake Ontario, Canada and their potential preservation in the sediment record. *Environmental Pollution* 204: 17–25.
Cuthbert RN, Al-Jaibachi R, Dalu T, Dick JT, and Callaghan A (2019) The influence of microplastics on trophic interaction strengths and oviposition preferences of dipterans. *Science of the Total Environment* 651: 2420–2423.
De Falco F, Di Pace E, Cocca M, and Avella M (2019) The contribution of washing processes of synthetic clothes to microplastic pollution. *Scientific Reports* 9: 6633.
Derraik JG (2002) The pollution of the marine environment by plastic debris: A review. *Marine Pollution Bulletin* 44: 842–852.
Di M and Wang J (2018) Microplastics in surface waters and sediments of the Three Gorges Reservoir, China. *Science of the Total Environment* 616-617: 1620–1627.
Dris R, Gasperi J, Rocher V, et al. (2015) Microplastic contamination in an urban area: A case study in Greater Paris. *Environmental Chemistry* 12: 592.
Dris R, Gasperi J, Rocher V, and Tassin B (2018) Synthetic and non-synthetic anthropogenic fibers in a river under the impact of Paris Megacity: Sampling methodological aspects and flux estimations. *Science of the Total Environment* 618: 157–164.
Duis K and Coors A (2016) Microplastics in the aquatic and terrestrial environment: Sources (with a specific focus on personal care products), fate and effects. *Environmental Sciences Europe* 28: 1–25.
Dümichen E, Eisentraut P, Bannick CG, et al. (2017) Fast identification of microplastics in complex environmental samples by a thermal degradation method. *Chemosphere* 174: 572–584.
Eerkes-Medrano D and Thompson R (2018) Occurrence, fate, and effect of microplastics in freshwater systems. In: Zeng EY (ed.) *Microplastic contamination in aquatic environments*, pp. 95–132. Elsevier.
Ehlers S, Manz W, and Koop J (2019) Microplastics of different characteristics are incorporated into the larval cases of the freshwater caddisfly *Lepidostoma basale*. *Aquatic Biology* 28: 67–77.
Eltemsah YS and Bøhn T (2019) Acute and chronic effects of polystyrene microplastics on juvenile and adult Daphnia magna. *Environmental Pollution* 254: 112919.
Eriksen M, Mason S, Wilson S, et al. (2013) Microplastic pollution in the surface waters of the Laurentian Great Lakes. *Marine Pollution Bulletin* 77: 177–182.
Estahbanati S and Fahrenfeld N (2016) Influence of wastewater treatment plant discharges on microplastic concentrations in surface water. *Chemosphere* 162: 277–284.
Plastics Europe (2019) *Plastics-the facts. An analysis of European plastics production, demand and waste data*. Plastics Europe: Association of Plastic Manufacturers.
de Faria E, Girard P, Nardes CS, et al. (2019) Microplastics pollution in the South American Pantanal. *PeerJ Preprints* 7. e27754v1.
Faure F, Corbaz M, and Baecher H (2012) Pollution due to plastics and microplastics in Lake Geneva and in the Mediterranean Sea. *Archives Des Sciences* 65: 157–164.
Faure F, Demars C, Wieser O, Kunz M, and de Alencastro LF (2015) Plastic pollution in Swiss surface waters: Nature and concentrations, interaction with pollutants. *Environmental Chemistry* 12: 582.
Fischer EK, Paglialonga L, Czech E, and Tamminga M (2016) Microplastic pollution in lakes and lake shoreline sediments—A case study on Lake Bolsena and Lake Chiusi (central Italy). *Environmental Pollution* 213: 648–657.
Free CM, Jensen OP, Mason SA, et al. (2014) High-levels of microplastic pollution in a large, remote, mountain lake. *Marine Pollution Bulletin* 85: 156–163.
Frias JPG and Nash R (2019) Microplastics: Finding a consensus on the definition. *Marine Pollution Bulletin* 138: 145–147.
Frydkjær C, Iversen N, and Roslev P (2017) Ingestion and egestion of microplastics by the cladoceran *Daphnia magna*: Effects of regular and irregular shaped plastic and sorbed phenanthrene. *Bulletin of Environmental Contamination and Toxicology* 99: 655–661.
GESAMP (2015). Sources, fate and effects of microplastics in the marine environment: A global assessment, Kershaw, P. J., (ed.). IMO/FAO/UNESCO-IOC/UNIDO/WMO/IAEA/UN/UNEP/UNDP Joint Group of Experts on the Scientific Aspects of Marine Environmental Protection. Rep. Stud. GESAMP **90**, 96 p.
Girão AV, Caputo G, and Ferro MC (2016) Application of scanning electron microscopy-energy dispersive X-ray spectroscopy (SEM.EDS). In: Rocha-Santos T and Duarte A (eds.) *Characterization and analysis of microplastics*. Elsevier.
Guilhermino L, Vieira LR, Ribeiro D, et al. (2018) Uptake and effects of the antimicrobial florfenicol, microplastics and their mixtures on freshwater exotic invasive bivalve *Corbicula fluminea*. *Science of the Total Environment* 622-623: 1131–1142.
Gündoğdu S, Çevik C, Güzel E, and Kilercioğlu S (2018) Microplastics in municipal wastewater treatment plants in Turkey: A comparison of the influent and secondary effluent concentrations. *Environmental Monitoring and Assessment* 190: 1–10.
Guo X and Wang J (2019) The chemical behaviors of microplastics in marine environment: A review. *Marine Pollution Bulletin* 142: 1–14.

Haghi BN and Banaee M (2017) Effects of micro-plastic particles on paraquat toxicity to common carp (*Cyprinus carpio*): Biochemical changes. *International Journal of Environmental Science and Technology* 14(3): 521–530.

Hahladakis JN, Velis CA, Weber R, Iacovidou E, and Purnell P (2018) An overview of chemical additives present in plastics: Migration, release, fate and environmental impact during their use, disposal and recycling. *Journal of Hazardous Materials* 344: 179–199.

Hartmann N, Hüffer T, Thompson RC, et al. (2019) Are we speaking the same language? Recommendations for a definition and categorization framework for plastic debris. *Environmental Science and Technology* 53: 1039–1047.

Hendrickson E, Minor EC, and Schreiner K (2018) Microplastic abundance and composition in Western Lake Superior as determined via microscopy, Pyr-GC/MS, and FTIR. *Environmental Science and Technology* 52: 1787–1796.

Horton AA, Svendsen C, Williams RJ, Spurgeon DJ, and Lahive E (2017) Large microplastic particles in sediments of tributaries of the River Thames, UK—Abundance, sources and methods for effective quantification. *Marine Pollution Bulletin* 114: 218–226.

Horton AA, Jürgens MD, Lahive E, van Bodegom PM, and Vijver MG (2018) The influence of exposure and physiology on microplastic ingestion by the freshwater fish *Rutilus rutilus* (roach) in the River Thames, UK. *Environmental Pollution* 236: 188–194.

Hu L, Chernick M, Hinton DE, and Shi H (2018) Microplastics in small waterbodies and tadpoles from Yangtze River Delta, China. *Environmental Science and Technology* 52: 8885–8893.

Hüffer T and Hofmann T (2016) Sorption of non-polar organic compounds by micro-sized plastic particles in aqueous solution. *Environmental Pollution* 214: 194–201.

Hurley RR, Woodward JC, and Rothwell JJ (2017) Ingestion of microplastics by freshwater *Tubifex* worms. *Environmental Science and Technology* 51: 12844–12851.

Hurley R, Woodward J, and Rothwell JJ (2018) Microplastic contamination of river beds significantly reduced by catchment-wide flooding. *Nature Geoscience* 11: 251–257.

Imhof HK, Ivleva NP, Schmid J, Niessner R, and Laforsch C (2013) Contamination of beach sediments of a subalpine lake with microplastic particles. *Current Biology* 23: R867–R868.

Jabeen K, Su L, Li J, et al. (2017) Microplastics and mesoplastics in fish from coastal and fresh waters of China. *Environmental Pollution* 221: 141–149.

Jaikumar G, Baas J, Brun NR, Vijver MG, and Bosker T (2018) Acute sensitivity of three cladoceran species to different types of microplastics in combination with thermal stress. *Environmental Pollution* 239: 733–740.

Jaikumar G, Brun NR, Vijver MG, and Bosker T (2019) Reproductive toxicity of primary and secondary microplastics to three cladocerans during chronic exposure. *Environmental Pollution* 249: 638–646.

Jambeck JR, Geyer R, Wilcox C, et al. (2015) Plastic waste inputs from land into the ocean. *Science* 347: 768–771.

Jemec A, Horvat P, Kunej U, Bele M, and Kržan A (2016) Uptake and effects of microplastic textile fibers on freshwater crustacean *Daphnia magna*. *Environmental Pollution* 219: 201–209.

Jiang C, Yin L, Wen X, et al. (2018) Microplastics in sediment and surface water of west Dongting Lake and South Dongting Lake: Abundance, source and composition. *International Journal of Environmental Research and Public Health* 15: 2164.

Jiang C, Yin L, Li Z, et al. (2019) Microplastic pollution in the rivers of the Tibet Plateau. *Environmental Pollution* 249: 91–98.

Jin Y, Xia J, Pan Z, et al. (2018) Polystyrene microplastics induce microbiota dysbiosis and inflammation in the gut of adult zebrafish. *Environmental Pollution* 235: 322–329.

Kalčíková G, Alič B, Skalar T, Bundschuh M, and Gotvajn AŽ (2017) Wastewater treatment plant effluents as source of cosmetic polyethylene microbeads to freshwater. *Chemosphere* 188: 25–31.

Kapp KJ and Yeatman E (2018) Microplastic hotspots in the snake and lower Columbia rivers: A journey from the Greater Yellowstone Ecosystem to the Pacific Ocean. *Environmental Pollution* 241: 1082–1090.

Käppler A, Fischer D, Oberbeckmann S, et al. (2016) Analysis of environmental microplastics by vibrational microspectroscopy: FTIR, Raman or both? *Analytical and Bioanalytical Chemistry* 408: 8377–8391.

Karami A, Romano N, Galloway T, and Hamzah H (2016) Virgin microplastics cause toxicity and modulate the impacts of phenanthrene on biomarker responses in African catfish (*Clarias gariepinus*). *Environmental Research* 151: 58–70.

Khan FR, Syberg K, Shashoua Y, and Bury NR (2015) Influence of polyethylene microplastic beads on the uptake and localization of silver in zebrafish (*Danio rerio*). *Environmental Pollution* 206: 73–79.

Klein S, Worch E, and Knepper TP (2015) Occurrence and spatial distribution of microplastics in river shore sediments of the Rhine-main area in Germany. *Environmental Science and Technology* 49: 6070–6076.

Kleinteich J, Seidensticker S, Marggrander N, and Zarfl C (2018) Microplastics reduce short-term effects of environmental contaminants. Part II: Polyethylene particles decrease the effect of polycyclic aromatic hydrocarbons on microorganisms. *International Journal of Environmental Research and Public Health* 15: 287.

Koelmans AA, Bakir A, Burton GA, and Janssen CR (2016) Microplastic as a vector for chemicals in the aquatic environment: Critical review and model-supported reinterpretation of empirical studies. *Environmental Science and Technology* 50: 3315–3326.

Lagarde F, Olivier O, Zanella M, et al. (2016) Microplastic interactions with freshwater microalgae: Hetero-aggregation and changes in plastic density appear strongly dependent on polymer type. *Environmental Pollution* 215: 331–339.

Lahens L, Strady E, Kieu-Le T-C, et al. (2018) Macroplastic and microplastic contamination assessment of a tropical river (Saigon River, Vietnam) transversed by a developing megacity. *Environmental Pollution* 236: 661–671.

Lares M, Ncibi MC, Sillanpää M, and Sillanpää M (2018) Occurrence, identification and removal of microplastic particles and fibers in conventional activated sludge process and advanced MBR technology. *Water Research* 133: 236–246.

Lei L, Wu S, Lu S, et al. (2018) Microplastic particles cause intestinal damage and other adverse effects in zebrafish *Danio rerio* and nematode *Caenorhabditis elegans*. *Science of the Total Environment* 619–620: 1–8.

Leslie H, Brandsma S, van Velzen MJ, and Vethaak A (2017) Microplastics en route: Field measurements in the Dutch river delta and Amsterdam canals, wastewater treatment plants, North Sea sediments and biota. *Environment International* 101: 133–142.

Li L, Li Q, Su L, et al. (2019) The uptake of microfibers by freshwater Asian clams (*Corbicula fluminea*) varies based upon physicochemical properties. *Chemosphere* 221: 107–114.

Lin L, Zuo L-Z, Peng J-P, et al. (2018) Occurrence and distribution of microplastics in an urban river: A case study in the Pearl River along Guangzhou City, China. *Science of the Total Environment* 644: 375–381.

Liu F, Olesen KB, Borregaard AR, and Vollertsen J (2019) Microplastics in urban and highway stormwater retention ponds. *Science of the Total Environment* 671: 992–1000.

Lu Y, Zhang Y, Deng Y, et al. (2016) Uptake and accumulation of polystyrene microplastics in zebrafish (*Danio rerio*) and toxic effects in liver. *Environmental Science and Technology* 50: 4054–4060.

Lv W, Lv W, Zhou W, et al. (2019) Microplastic pollution in rice-fish co-culture system: A report of three farmland stations in Shanghai, China. *Science of the Total Environment* 652: 1209–1218.

Ma Y, Huang A, Cao S, et al. (2016) Effects of nanoplastics and microplastics on toxicity, bioaccumulation, and environmental fate of phenanthrene in fresh water. *Environmental Pollution* 219: 166–173.

Maes T, Jessop R, Wellner N, Haupt K, and Mayes AG (2017) A rapid-screening approach to detect and quantify microplastics based on fluorescent tagging with Nile Red. *Scientific reports* 7: 44501.

Magni S, Gagné F, André C, et al. (2018) Evaluation of uptake and chronic toxicity of virgin polystyrene microbeads in freshwater zebra mussel *Dreissena polymorpha* (Mollusca: Bivalvia). *Science of the Total Environment* 631-632: 778–788.

Mahon AM, O'Connell B, Healy MG, et al. (2017) Microplastics in sewage sludge: Effects of treatment. *Environmental Science and Technology* 51: 810.

Mani T, Hauk A, Walter U, and Burkhardt-Holm P (2015) Microplastics profile along the Rhine River. *Scientific Reports* 5: 17988.

Mason SA, Garneau D, Sutton R, et al. (2016a) Microplastic pollution is widely detected in US municipal wastewater treatment plant effluent. *Environmental Pollution* 218: 1045–1054.

Mason SA, Kammin L, Eriksen M, et al. (2016b) Pelagic plastic pollution within the surface waters of Lake Michigan, USA. *Journal of Great Lakes Research* 42: 753–759.
Mateos-Cárdenas A, Scott DT, Seitmaganbetova G, et al. (2019) Polyethylene microplastics adhere to *Lemna minor* (L.), yet have no effects on plant growth or feeding by *Gammarus duebeni* (Lillj.). *Science of the Total Environment* 689: 413–421.
Mato Y, Isobe T, Takada H, et al. (2001) Plastic resin pellets as a transport medium for toxic chemicals in the marine environment. *Environmental Science and Technology* 35: 318–324.
McCormick A, Hoellein TJ, Mason SA, Schluep J, and Kelly JJ (2014) Microplastic is an abundant and distinct microbial habitat in an urban river. *Environmental Science and Technology* 48: 11863.
McCormick AR, Hoellein TJ, London MG, et al. (2016) Microplastic in surface waters of urban rivers: Concentration, sources, and associated bacterial assemblages. *Ecosphere* 7: e01556.
McNeish RE, Kim LH, Barrett HA, et al. (2018) Microplastic in riverine fish is connected to species traits. *Scientific Reports* 8: 11639.
Michielssen MR, Michielssen ER, Ni J, and Duhaime MB (2016) Fate of microplastics and other small anthropogenic litter (SAL) in wastewater treatment plants depends on unit processes employed. *Environmental Science: Water Research and Technology* 2: 1064–1073.
Miller RZ, Watts AJR, Winslow BO, Galloway TS, and Barrows APW (2017) Mountains to the sea: River study of plastic and non-plastic microfiber pollution in the northeast USA. *Marine Pollution Bulletin* 124: 245–251.
Mintenig S, Int-Veen I, Löder MG, Primpke S, and Gerdts G (2017) Identification of microplastic in effluents of waste water treatment plants using focal plane array-based micro-Fourier-transform infrared imaging. *Water Research* 108: 365–372.
Mintenig SM, Bäuerlein PS, Koelmans AA, Dekker SC, and van Wezel AP (2018) Closing the gap between small and smaller: Towards a framework to analyse nano- and microplastics in aqueous environmental samples. *Environmental Science: Nano* 5(7): 1640–1649.
Monneret C (2017) What is an endocrine disruptor? *Comptes Rendus Biologies* 340: 403–405.
Moore C, Lattin G, and Zellers A (2011) Quantity and type of plastic debris flowing from two urban rivers to coastal waters and beaches of Southern California. *Journal of Integrated Coastal Zone Management* 11: 65–73.
Murphy F and Quinn B (2018) The effects of microplastic on freshwater *Hydra attenuata* feeding, morphology & reproduction. *Environmental Pollution* 234: 487–494.
Murphy F, Ewins C, Carbonnier F, and Quinn B (2016) Wastewater treatment works (WwTW) as a source of microplastics in the aquatic environment. *Environmental Science & Technology* 50: 5800–5808.
Napper IE and Thompson RC (2016) Release of synthetic microplastic plastic fibres from domestic washing machines: Effects of fabric type and washing conditions. *Marine Pollution Bulletin* 112: 39–45.
Nel HA, Dalu T, and Wasserman RJ (2018) Sinks and sources: Assessing microplastic abundance in river sediment and deposit feeders in an Austral temperate urban river system. *Science of the Total Environment* 612: 950–956.
Ogonowski M, Schür C, Jarsén Å, and Gorokhova E (2016) The effects of natural and anthropogenic microparticles on individual fitness in *Daphnia magna*. *PLoS One* 11: e0155063.
Olesen KB, Stephansen DA, van Alst N, and Vollertsen J (2019) Microplastics in a stormwater pond. *Water* 11: 1466.
Ou H and Zang EY (2018) Occurrence and fate of microplastics in wastewater treatment plants. In: Zeng EY (ed.) *Microplastic contamination in aquatic environments. An emerging matter of environmental urgency*. Elsevier.
Panko JM, Chu J, Kreider ML, and Unice KM (2013) Measurement of airborne concentrations of tire and road wear particles in urban and rural areas of France, Japan, and the United States. *Atmospheric Environment* 72: 192–199.
Peng G, Xu P, Zhu B, Bai M, and Li D (2018) Microplastics in freshwater river sediments in Shanghai, China: A case study of risk assessment in mega-cities. *Environmental Pollution* 234: 448–456.
Peters CA and Bratton SP (2016) Urbanization is a major influence on microplastic ingestion by sunfish in the Brazos River Basin, Central Texas, USA. *Environmental Pollution* 210: 380–387.
Phillips MB and Bonner TH (2015) Occurrence and amount of microplastic ingested by fishes in watersheds of the Gulf of Mexico. *Marine Pollution Bulletin* 100: 264–269.
Prata JC, da Costa JP, Duarte AC, and Rocha-Santos T (2019a) Methods for sampling and detection of microplastics in water and sediment: A critical review. *Trend in Analytical Chemistry* 110: 150–159.
Prata JC, da Costa JP, Lopes I, Duarte AC, and Rocha-Santos T (2019b) Environmental exposure to microplastics: An overview on possible human health effects. *Science of the Total Environment* 702: 134455.
Redondo Hasselerharm PE, Falahudin D, Peeters E, and Koelmans AA (2018) Microplastic effect thresholds for freshwater benthic macroinvertebrates. *Environmental Science and Technology* 52: 2278–2286.
Reed C (2015) Dawn of the plasticene age. *New Scientist* 225: 28–32.
Rehse S, Kloas W, and Zarfl C (2016) Short-term exposure with high concentrations of pristine microplastic particles leads to immobilisation of *Daphnia magna*. *Chemosphere* 153: 91–99.
Rehse S, Kloas W, and Zarfl C (2018) Microplastics reduce short-term effects of environmental contaminants. Part I: Effects of bisphenol A on freshwater zooplankton are lower in presence of polyamide particles. *International Journal of Environmental Research and Public Health* 15: 280.
Rios Mendoza LM (2019) *Microplastic particles St. Louis River Estuary and Lake Superior. Large Lakes Research. Conference International Association for Great Lakes Research. June 10–14, Brockport, NY.* .
Rios Mendoza LM and Balcer M (2019a) Microplastics in freshwater environments: A review of quantification assessment. *Trends in Analytical Chemistry* 113: 402–408.
Rios Mendoza LM and Balcer M (2019b) Association of hazardous compounds with microplastics in freshwater ecosystems. In: Karapanagioti HK and Kalavrouziotis IK (eds.) *Microplastics in water and wastewater*. London, UK: International Water Association, IWA Publishing.
Rios LM, Moore C, and Jones PR (2007) Persistent organic pollutants carried by synthetic polymers in the ocean environment. *Marine Pollution Bulletin* 54: 1230–1237.
Rist S, Baun A, and Hartmann NB (2017) Ingestion of micro- and nanoplastics in *Daphnia magna*—Quantification of body burdens and assessment of feeding rates and reproduction. *Environmental Pollution* 228: 398–407.
Roch S and Brinker A (2017) Rapid and efficient method for the detection of microplastic in the gastrointestinal tract of fishes. *Environmental Science and Technology* 51: 4522–4530.
Roch S, Walter T, Ittner LD, Friedrich C, and Brinker A (2019) A systematic study of the microplastic burden in freshwater fishes of south-western Germany—Are we searching at the right scale? *Science of the Total Environment* 689: 1001–1011.
Rochman CM, Browne MA, Halpern BS, et al. (2013a) Policy: Classify plastic waste as hazardous. *Nature* 494: 169–171.
Rochman CM, Hoh E, Kurobe T, and Teh SJ (2013b) Ingested plastic transfers hazardous chemicals to fish and induces hepatic stress. *Scientific Reports* 3: 3263.
Rochman CM, Parnis JM, Browne MA, et al. (2017) Direct and indirect effects of different types of microplastics on freshwater prey (*Corbicula fluminea*) and their predator (*Acipenser transmontanus*). *PloS One* 12: e0187664.
Rodrigues M, Abrantes N, Gonçalves FJ, Gonçalves AM, Nogueira H, and Marques J (2018) Spatial and temporal distribution of microplastics in water and sediments of a freshwater system (Antuã River, Portugal). *Science of the Total Environment* 633: 1549–1559.
Rodrigues JP, Duearte AC, Santos-Echeandia J, and Rocha-Santos T (2019) Significance of interactions between microplastics and POPs in the marine environment: A critical overview. *Trends in Analytical Chemistry* 111: 252–260.
Rodríguez-Seijo A and Pereira R (2017) Morphological and physical characterization of microplastics. In: Rocha-Santos TAP and Duarte AC (eds.) *Comprehensive analytical chemistry*, pp. 49–66. Amsterdam, Netherlands: Elsevier.
Rosenkranz P, Chaudhry Q, Stone V, and Fernandes TF (2009) A comparison of nanoparticle and fine particle uptake by *Daphnia magna*. *Environmental Toxicology and Chemistry* 28(10): 2142–2149.

Ryan PG, Moore CJ, van Franeker JA, and Moloney CL (2009) Monitoring the abundance of plastic debris in the marine environment. *Philosophical Transactions of the Royal Society of London. Series B: Biological Sciences* 364: 1999–2012.

Sanchez W, Bender C, and Porcher J-M (2014) Wild gudgeons (*Gobio gobio*) from French rivers are contaminated by microplastics: Preliminary study and first evidence. *Environmental Research* 128: 98–100.

Schmidt LK, Bochow M, Imhof HK, and Oswald SE (2018) Multi-temporal surveys for microplastic particles enabled by a novel and fast application of SWIR imaging spectroscopy—Study of an urban watercourse traversing the city of Berlin. Germany. *Environmental Pollution* 239: 579–589.

Schür C, Rist S, Baun A, et al. (2019) When fluorescence is not a particle: The tissue translocation of microplastics in *Daphnia magna* seems an artifact. *Environmental Toxicology and Chemistry* 38: 1495–1503.

Serranti S, Palmieri R, Bonifazi G, and Cózar A (2018) Characterization of microplastic litter from oceans by an innovative approach based on hyperspectral imaging. *Waste Management* 76: 117–125.

Shruti V, Jonathan M, Rodriguez-Espinosa P, and Rodríguez-González F (2019) Microplastics in freshwater sediments of Atoyac River basin, Puebla City, Mexico. *Science of the Total Environment* 654: 154–163.

Sighicelli M, Pietrelli L, Lecce F, et al. (2018) Microplastic pollution in the surface waters of Italian Subalpine Lakes. *Environmental Pollution* 236: 645–651.

Silva-Cavalcanti JS, Silva JDB, de França EJ, de Araújo MCB, and Gusmão F (2017) Microplastics ingestion by a common tropical freshwater fishing resource. *Environmental Pollution* 221: 218–226.

Simon M, van Alst N, and Vollertsen J (2018) Quantification of microplastic mass and removal rates at wastewater treatment plants applying Focal Plane Array (FPA)-based Fourier Transform Infrared (FT-IR) imaging. *Water Research* 142: 1–9.

Sleight VA, Bakir A, Thompson RC, and Henry TB (2017) Assessment of microplastic-sorbed contaminant bioavailability through analysis of biomarker gene expression in larval zebrafish. *Marine Pollution Bulletin* 116(1–2): 291–297.

Slootmaekers B, Cartarcy Carteny C, Belpaire C, et al. (2019) Microplastic contamination in gudgeons (*Gobio gobio*) from Flemish rivers (Belgium). *Environmental Pollution* 244: 675–684.

Sommer F, Dietze V, Baum A, et al. (2018) Tire abrasion as a major source of microplastics in the environment. *Aerosol and Air Quality Research* 18: 2014–2028.

Sruthy S and Ramasamy E (2017) Microplastic pollution in Vembanad Lake, Kerala, India: The first report of microplastics in lake and estuarine sediments in India. *Environmental Pollution* 222: 315–322.

Straub S, Hirsch PE, and Burkhardt-Holm P (2017) Biodegradable and petroleum-based microplastics do not differ in their ingestion and excretion but in their biological effects in a freshwater invertebrate *Gammarus fossarum*. *International Journal of Environmental Research and Public Health* 14: 774.

Su L, Xue Y, Li L, et al. (2016) Microplastics in Taihu Lake, China. *Environmental Pollution* 216: 711–719.

Su L, Cai H, Kolandhasamy P, et al. (2018) Using the Asian clam as an indicator of microplastic pollution in freshwater ecosystems. *Environmental Pollution* 234: 347–355.

Su L, Nan B, Hassell KL, Craig NJ, and Pettigrove V (2019) Microplastics biomonitoring in Australian urban wetlands using a common noxious fish (*Gambusia holbrooki*). *Chemosphere* 228: 65–74.

Talvitie J, Mikola A, Setälä O, Heinonen M, and Koistinen A (2017) How well is microlitter purified from wastewater?—A detailed study on the stepwise removal of microlitter in a tertiary level wastewater treatment plant. *Water Research* 109: 164–172.

Tan X, Yu X, Cai L, Wang J, and Peng J (2019) Microplastics and associated PAHs in surface water from the Feilaixia Reservoir in the Beijiang River, China. *Chemosphere* 221: 834–840.

Thompson RC, Swan SH, Moore CJ, and Saal FS (2009) Our plastic age. *Philosophical Transactions of the Royal Society B: Biological Sciences* 364: 1973–1976.

Tibbetts J, Krause S, Lynch I, and Sambrook Smith G (2018) Abundance, distribution, and drivers of microplastic contamination in urban river environments. *Water* 10: 1597.

Toumi H, Abidli S, and Bejaoui M (2019) Microplastics in freshwater environment: The first evaluation in sediments from seven water streams surrounding the lagoon of Bizerte (Northern Tunisia). *Environmental Science and Pollution Research International* 26: 14673–14682.

Townsend KR, Lu HC, Sharley DJ, and Pettigrove V (2019) Associations between microplastic pollution and land use in urban wetland sediments. *Environmental Science and Pollution Research International* 26: 22551–22561.

United Nations. (2015). Transforming our world: The 2030 Agenda for Sustainable Development. A/Res/70/1.

Utracki LA (2003) *Polymer blends handbook*. USA: Kluwer Academic Publishers.

Vermaire JC, Pomeroy C, Herczegh SM, Haggart O, and Murphy M (2017) Microplastic abundance and distribution in the open water and sediment of the Ottawa River, Canada, and its tributaries. *Facets* 2: 301–314.

Vijayan A and Mohan GS (2013) Prediction of effluent treatment plant performance in a diary industry using artificial neural network technique. *International Journal of Science and Research* 5: 10.

Wagner M and Lambert S (eds.) (2018) *Freshwater microplastics. Emerging environmental contaminants?*. In: *The handbook of environmental chemistry*In: Springer.

Wagner J, Wang Z-M, Ghosal S, et al. (2017) Novel method for the extraction and identification of microplastics in ocean trawl and fish gut matrices. *Analytical Methods* 9: 1479–1490.

Wang J and Wang W (2018) Investigation of microplastics in aquatic environments: An overview of the methods used, from field sampling to laboratory analysis. *Trends in Analytical Chemistry* 108: 195–202.

Wang J, Peng J, Tan Z, et al. (2017a) Microplastics in the surface sediments from the Beijiang River littoral zone: Composition, abundance, surface textures and interaction with heavy metals. *Chemosphere* 171: 248–258.

Wang J, Wang W, Ndungu AW, and Li Z (2017b) Microplastics pollution in inland freshwaters of China: A case study in urban surface waters of Wuhan, China. *Science of the Total Environment* 575: 1369–1374.

Wang J, Wang W, Yuan W, and Chen Y (2018) Microplastics in surface waters of Dongting Lake and Hong Lake, China. *Science of the Total Environment* 633: 539–545.

Wang J, Liu X, and Liu G (2019a) Sorption behavior of phenanthrene, nitrobenzene, and naphthalene on mesoplastics and microplastics. *Environmental Science and Pollution Research International* 26: 12563–12573.

Wang Z, Qin Y, Li W, et al. (2019b) Microplastic contamination in freshwater: First observation in Lake Ulansuhai, Yellow River Basin, China. *Environmental Chemistry Letters* 1–10.

Wardrop P, Shimeta J, Nugegoda D, et al. (2016) Chemical pollutants sorbed to ingested microbeads from personal care products accumulate in fish. *Environmental Science and Technology* 50: 4037–4044.

Weber A, Scherer C, Brennholt N, Reifferscheid G, and Wagner M (2018) PET microplastics do not negatively affect the survival, development, metabolism and feeding activity of the freshwater invertebrate *Gammarus pulex*. *Environmental Pollution* 234: 181–189.

Wen B, Zhang N, Jin SR, et al. (2017) Microplastics have a more profound impact than elevated temperatures on the predatory performance, digestion and energy metabolism of an Amazonian cichlid. *Aquatic Toxicology (Amsterdam, Netherlands)* 195: 67.

Windsor FM, Tilley RM, Tyler CR, and Ormerod SJ (2019) Microplastic ingestion by riverine macroinvertebrates. *Science of the Total Environment* 646: 68–74.

Xiong X, Zhang K, Chen X, et al. (2018) Sources and distribution of microplastics in China's largest inland lake—Qinghai Lake. *Environmental Pollution* 235: 899–906.

Xu J, Xu B, Liu F, and Brookes PC (2018) The sorption kinetics and isotherms of sulfamethoxazole with polyethylene microplastics. *Marine Pollution Bulletin* 131: 191–196.

Yin L, Jiang C, Wen X, et al. (2019) Microplastic pollution in surface Water of Urban Lakes in Changsha, China. *International Journal of Environmental Research and Public Health* 16: 1650.

Zbyszewski M and Corcoran P (2011) Distribution and degradation of fresh water plastic particles along the beaches of Lake Huron, Canada. *Water, Air, & Soil Pollution* 220: 365–372.

Zbyszewski M, Corcoran PL, and Hockin A (2014) Comparison of the distribution and degradation of plastic debris along shorelines of the Great Lakes, North America. *Journal of Great Lakes Research* 40: 288–299.

Zettler ER, Mincer TJ, and Amaral-Zettler LA (2013) Life in the "plastisphere": Microbial communities on plastic marine debris. *Environmental Science and Technology* 47: 7137.
Zhang K, Gong W, Lv J, Xiong X, and Wu C (2015) Accumulation of floating microplastics behind the Three Gorges Dam. *Environmental Pollution* 204: 117–123.
Zhang K, Su J, Xiong X, et al. (2016) Microplastic pollution of lakeshore sediments from remote lakes in Tibet plateau, China. *Environmental Pollution* 219: 450–455.
Zhang K, Xiong X, Hu H, et al. (2017) Occurrence and characteristics of microplastic pollution in Xiangxi Bay of Three Gorges Reservoir, China. *Environmental Science and Technology* 51: 3794–3801.
Zhang P, Yan Z, Lu G, and Ji Y (2019a) Single and combined effects of microplastics and roxithromycin on *Daphnia magna*. *Environmental Science and Pollution Research* 26: 17010–17020.
Zhang X, Leng Y, Liu X, Huang K, and Wang J (2019b) Microplastics' pollution and risk assessment in an urban river: A case study in the Yongjiang River, Nanning City, South China. *Exposure and Health* 1–11.
Zheng K, Fan Y, Zhu Z, et al. (2019) Occurrence and Species-specific distribution of plastic Debris in wild freshwater fish from the Pearl River Catchment, China. *Environmental Toxicology and Chemistry* 38: 1504–1513.
Ziajahromi S (2018) *Identification and quantification of microplastics in wastewater treatment plant effluent: Investigation of the fate and biological effects*. Queensland, Australia: Griffith University.
Ziajahromi S, Kumar A, Neale PA, and Leusch FDL (2017a) Impact of microplastic beads and fibers on waterflea (*Ceriodaphnia dubia*) survival, growth, and reproduction: Implications of single and mixture exposures. *Environmental Science and Technology* 51: 13397–13406.
Ziajahromi S, Neale PA, Rintoul L, and Leusch FDL (2017b) Wastewater treatment plants as a pathway for microplastics: Development of a new approach to sample wastewater-based microplastics. *Water Research* 112: 93–99.
Ziajahromi S, Kumar A, Neale PA, and Leusch FDL (2018) Environmentally relevant concentrations of polyethylene microplastics negatively impact the survival, growth and emergence of sediment-dwelling invertebrates. *Environmental Pollution* 236: 425–431.
Zocchi M and Sommaruga R (2019) Microplastics modify the toxicity of glyphosate on *Daphnia magna*. *Science of the Total Environment* 697: 134194.

Further Reading

Guo X and Wang J (2019) The chemical behaviors of microplastics in marine environment: A review. *Marine Pollution Bulletin* 142: 1–14.
Koelmans AA, Mohamed Nor NH, Hermsen E, et al. (2019) Microplastics in freshwaters and drinking water: Critical review and assessment of data quality. *Water Research* 155: 410–422.
Roch S, Walter T, Ittner LD, Friedrich C, and Brinker A (2019) A systematic study of the microplastic burden in freshwater fishes of south-western Germany—Are we searching at the right scale? *Science of the Total Environment* 689: 1001–1011.
Rocha-Santos TA and Duarte AC (eds.) (2017) *Comprehensive analytical chemistry 75: Characterization and analysis of microplastics*. Amsterdam, Netherlands: Elsevier.
Schür C, Rist S, Baun A, et al. (2019) When fluorescence is not a particle: The tissue translocation of microplastics in *Daphnia magna* seems an artifact. *Environmental Toxicology and Chemistry* 38: 1495–1503.
Zeng EY (ed.) (2018) *Microplastic contamination in aquatic environments. An emerging matter of environmental urgency*. Amsterdam, Netherlands: Elsevier.