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REVIEW ARTICLE

Microplastic pollution in riverine ecosystems: threats posed on macroinvertebrates

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Abstract

Microplastics (MPs) are pollutants of emerging concern that have been reported in terrestrial and aquatic ecosystems as well as in food items. The increasing production and use of plastic materials have led to a rise in MP pollution in aquatic ecosystems. This review aimed at providing an overview of the abundance and distribution of MPs in riverine ecosystems and the potential effects posed on macroinvertebrates. Microplastics in riverine ecosystems are reported in all regions, with less research in Africa, South America, and Oceania. The abundance and distribution of MPs in riverine ecosystems are mainly affected by population density, economic activities, seasons, and hydraulic regimes. Ingestion of MPs has also been reported in riverine macroinvertebrates and has been incorporated in caddisflies cases. Further, bivalves and chironomids have been reported as potential indicators of MPs in aquatic ecosystems due to their ability to ingest MPs relative to environmental concentration. Fiber and fragments are the most common types reported. Meanwhile, polyethylene, polypropylene, polystyrene, polyethylene terephthalate (polyester), polyamide, and polyvinyl chloride are the most common polymers. These MPs are from materials/polymers commonly used for packaging, shopping/carrier bags, fabrics/textiles, and construction. Ingestion of MPs by macroinvertebrates can physically harm and inhibit growth, reproduction, feeding, and moulting, thus threatening their survival. In addition, MP ingestion can trigger enzymatic changes and cause oxidative stress in the organisms. There is a need to regulate the production and use of plastic materials, as well as disposal of the wastes to reduce MP pollution in riverine ecosystems.

Keywords Distribution · Fibers · Fragments · Ingestion · Oxidative stress · Physical harm · Polymer type

Introduction

Microplastics (MPs) are plastic materials measuring less than 5 mm at their longest dimension (GESAMP 2015; Lusher et al. 2017). Plastics are a group of synthetic and semisynthetic polymers derived from fossil sources (e.g., crude oil, coal) and organic materials, such as salt, cellulose, renewable compounds like starch, seaweed, palm, and grains (Lusher et al. 2017). Commonly used plastic materials include polyethylene (PE), polyvinyl chloride (PVC), polyester (PES) or polyethylene terephthalate (PET), acrylic, polystyrene (PS), polypropylene (PP), polyamide/nylon (PA), and elastane (Essel et al. 2015; Grigore 2017) and rubber (Lusher et al. 2017; Schell et al. 2020). Since the start of mass plastic production in the 1950s, MPs have been a global environmental challenge (Ostle et al. 2019; Dumbili and Henderson 2020; Xu et al. 2021).

Studies have reported MPs in different aquatic environment such as oceans, estuaries (Wicaksono et al. 2020; Zhou et al. 2021), mangrove (Khuyen et al. 2022), rivers, and lakes (Hurley et al. 2017; Huang et al. 2022). Indeed, MP pollution has become research of interest in riverine system of European countries (Ehlers et al. 2019; Kiss et al. 2021), Asia (Chinfak et al. 2021; Kameda et al. 2021), North America (Christensen et al. 2020; Corcoran et al. 2020), Africa (Toumi et al. 2019; Kundu et al. 2022), and Oceania (Dikareva and Simon 2019).

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Moreover, less-disturbed environments such as Antarctica have been found to have MP pollution as well (González-Pleiter et al. 2020).

Microplastic pollution is associated with the rising use of plastic materials by the increasing human population (Lusher et al. 2017). Indeed, plastics are considered a key indicator of the Anthropocene (current geological epoch), through formation of a geological layer (Zalasiewicz et al. 2016; Porta 2021). The use of plastic materials is rising due to their functional properties such as durability, strength, low face value cost, resistance to corrosion, as well as electric and thermal insulation (Lusher et al. 2017). Therefore, plastics are used to replace other materials such as glass, metal, wood, cotton, and stones in making common goods such as utensils, packaging materials, carrier bags, clothing, and tanks (Shahnawaz et al. 2019). Between 1950 and 2015, about 6300 metric tons of plastic wastes was generated globally, of which, 12% were incinerated, 9% recycled, and the rest of the waste was buried or accumulated in landfills or the natural environment (Geyer et al. 2017). Currently, plastic materials make up more than 10% of solid waste in over 61 countries (Lusher et al. 2017). By 2050, if the current trend of plastic waste production and management does not change, about 12,000 metric tons of plastic wastes is expected to be in natural environments or landfills (Geyer et al. 2017).

Based on their source, MPs are categorized either as "primary" or "secondary" (Shahnawaz et al. 2019). Primary MPs are industrially produced within the size range below 5 mm, e.g., pre-production resin pellets and microbeads used in cosmetics, toothpaste, and soap (Boucher and Friot 2017), as well as for abrasive blasting (Derraik 2002). Meanwhile, secondary MPs result from the fragmentation and weathering of larger plastic materials, such as textiles and tires through physical, chemical, and biological degradation (Gallitelli et al. 2021). A wide range of MP particles have been detected in aquatic environments (Akindele et al. 2020; Campanale et al. 2020b), soils (Rillig et al. 2017; Du et al. 2020), atmosphere (Prata 2018), drinking water (Pivokonsky et al. 2018), food (Waring et al. 2018), and human blood (Leslie et al. 2022).

The introduction of MP particles into aquatic ecosystems poses great dangers to aquatic life and terrestrial organisms, including humans. Microplastics ingestion has been reported in aquatic organisms like fish (Lusher et al. 2013; Khan et al. 2020), birds (Lusher et al. 2017), and some macroinvertebrates (Hurley et al. 2017; Akindele et al. 2020). In macroinvertebrates, ingestion/absorption of MPs has been reported in dipterans (Nel et al. 2018; Akindele et al. 2020), oligochaetes (Hurley et al. 2017), ephemeropterans and trichopterans (Windsor et al. 2019), gastropods (Akindele et al. 2019), and bivalves (Berglund et al. 2019).

Ingestion/absorption of MPs by macroinvertebrates can cause internal damage and clog internal organs, such as the digestive and reproductive tracts, thus impairing feeding and reproduction, as well as growth and survival (Gago et al. 2020). In addition, the attachment of MPs onto appendages of organisms, such as limbs, affects their mobility, inhibiting feeding and increasing the risk of predation (Aljaibachi and Callaghan 2018). Once ingested, MPs can be transferred to the food web, alongside potentially harmful chemicals used as additives during plastic manufacture, such as brominated flame retardants, bisphenol A (BPA), phthalates, zinc borate, and alkylphenols (Ziccardi et al. 2016; Andrady 2017; Gago et al. 2020). In addition, MPs can transfer pollutants adsorbed from the environment, like polycyclic aromatic hydrocarbons (PAHs) to organisms (Andrady 2017; Gago et al. 2020). Chemical absorption by macroinvertebrates can cause hepatic stress and inflammation, leading to decreased growth rate (Setälä et al. 2016).

Pollution and the effects of MPs on aquatic life have widely been studied in marine environments and laboratory experiments (Lusher et al. 2017; Redondo-Hasselerharm et al. 2018a; Windsor et al. 2019). Unfortunately, less attention has been given to freshwater ecosystems, like lakes, rivers, and wetlands, despite their proximity to terrestrial sources of MP pollutants. Lotic ecosystems, such as rivers and streams, are important transportation systems of MPs into marine environments (Hamid et al. 2018; Rodrigues et al. 2018; Shahnawaz et al. 2019). Indeed, over 70% of marine MPs are estimated to emanate from freshwater sources (Garcia et al. 2021). As such, successful management of MP pollution in lentic ecosystems (e.g., seas and lakes) needs to begin from the sources (i.e., terrestrial environments) and transport systems (i.e., lotic/riverine ecosystems). Therefore, this provides summarized information on the potential sources, abundances, and global distribution of MPs in riverine ecosystems; MP ingestion/absorption by riverine macroinvertebrates; the effects of MP ingestion on macroinvertebrates; ecological and public health concerns associated with MP pollution; and the potential of macroinvertebrates as indicators of MP pollution in riverine ecosystems.

Materials and methods

A literature survey was conducted on peer review articles, dissertations/theses, and reports published between 2014 and November 2022. The keywords used included "microplastics," "microplastics in river/stream," "microplastics distribution," "microplastics ingestion/absorption in riverine macroinvertebrates," "effect of microplastics on freshwater macroinvertebrates," "caddisfly case building, microplastic," and "microplastics, public health." Academic/

scientific search engines such as Google scholar, Web of Science, Semantic Scholar, Scopus, and ResearchGate were the sources of data. The inclusion criteria were limited to the articles on MPs in riverine ecosystems and riverine macroinvertebrates. Articles on laboratory and field experiments on interactions (ingestion, case building for caddisflies, and burrowing) of freshwater macroinvertebrates with MPs and their effects were included. Articles aimed at guiding MP research, i.e., field sampling and laboratory extraction approaches, were not included. In addition, articles on MP pollution from riverine systems into the sea, or lake, and those that were lumped together (i.e., unspecified data for rivers in association with lakes/seas) were excluded.

Results and discussion

A total of 183 articles on MP pollution in riverine ecosystems were reviewed. Out of the 183 articles, 121, 84, and 21 articles reported on MP pollution in water, sediments, and macroinvertebrates, respectively (Table 1). It should be noted that some articles reported on more than one aspect, either water and sediment, water and macroinvertebrates, sediments and macroinvertebrates, or all aspects. Various terms for MP types were used by different researchers. In this article, MP types have been reported, i.e., fragment (fragment, particle, and flake), films (film and sheet); fiber (fiber, strip, rod-like, and line), pellet or bead (pellet, sphere, bead, and spherule), and foam (foam, PS foam, and styrofoam).

Sources, fate, and transportation of MPs into riverine ecosystems

Microplastic pollution in aquatic ecosystems originates largely from terrestrial environments, entering the water systems mainly through surface runoff and direct dumping, such as effluents and littering into the lotic systems (Hamid et al. 2018; van Emmerik and Schwarz 2020). Lotic systems further transport MPs to lentic systems such as lakes and oceans. Atmospheric transportation (and deposition) is another pathway of MPs into aquatic environments and could be responsible for the presence of MPs in pristine environments (González-Pleiter et al. 2020). Moreover, 80% of plastic pollution in the sea has been reported to originate from land (Eamrat et al. 2022). Factors that influence the fate and transport of MPs into aquatic ecosystems include climatic conditions, human activities, the type of the ecosystem, hydrological conditions, as well as MP density and source (Piccardo et al. 2021).

Currently, seven key sources of MPs have been identified globally, i.e., plastic pellets (pre-production pellets), synthetic textiles (fabrics), tires, road markings, marine coatings, personal care products (PCPs), and municipal dust (Boucher and Friot 2017). Plastic pellets (plastic powder, nibs, nurdles) mainly used as raw materials for manufacturing plastic products (e.g., carrier bags, utensils, basins) accidentally spill into the environment during transportation to factories, processing, and recycling (Lusher et al. 2017; Hann et al. 2018). Indeed, higher abundance of pellets have been reported near plastic manufacturing and recycling, due to losses during the manufacturing processes (Hidalgo-Ruz et al. 2012; Wicaksono et al. 2021). Meanwhile, fibers from synthetic textiles, such as PES, acrylic, and elastane, are lost during laundry, use, or are shed through wear and tear, and disposed of as wastes into water courses (Essel et al. 2015; Yang et al. 2019; Vassilenko et al. 2021). Wastewater treatment plants also release MPs, especially fibers and fragments into the water systems as effluents (Kay et al. 2018; Edo et al. 2020).

Tires are made from a mix of synthetic polymers like styrene butadiene rubber (SBR), natural rubber, and additives (Sundt et al. 2014; Schell et al. 2020). Abrasion of tires during driving leads to the unintentional release of tire dust which is either swept by wind or washed by urban runoff into the aquatic environments (Unice et al. 2013; Sundt et al. 2014). Thermoplastic marking paint is the commonest material used in road markings, such as zebra crossings, bumps, and dividers (Lassen et al. 2015; Horton et al. 2017). Road marking paints wear out due to weathering and abrasion by vehicles, hence released into the atmosphere and surface runoff, subsequently reaching water systems (Rhodes 2018).

On the other hand, marine coatings are used as anticorrosive or antifouling paintings on water vessels such as ships, submarines, and boats. They include polyurethane, vinyl, and lacquers (Boucher and Friot 2017) and are lost into the environment during coating application (by dripping), grounding, accidents, repair, and maintenance, as well as natural wearing/

Table 1 Summary of studies on MP type and polymer in water, sediments, and macroinvertebrates

Item	Water	Sediment	Macroinvertebrates
Reviewed studies (*n*)	121	84	21
Studies reporting MP type (%)	90.9	89.3	85.7
Most reported type	Fiber, fragment	Fiber, fragment	Fiber, fragment
Studies reporting MP polymer (%)	75	72.6	57.1
Most reported polymer	PE, PP, PS	PE, PP, PS	PES, PP, PA

peeling off (Turner 2021). In personal care products, such as cosmetics, toothpaste, and toiletries, microbeads are used at the sorbent phase for exfoliation, delivery of active ingredients, and enhancement of viscosity (Boucher and Friot 2017; Lei et al. 2017; Hoang et al. 2022). Microbeads are washed into wastewater during the application of these personal care products. Microplastic particles in municipal dust are lost during usage and maintenance of objects (e.g., footwear and utensils), city infrastructure (e.g., buildings and artificial turfs), as well as usage of abrasives and detergents (Magnusson et al. 2016).

The type of MP particles portrays a lot about the source or origin of the particles. Fibers, fragments, and film (sheet) are often secondary MP materials originating from the fragmentation of larger components during transportation, use, or after disposal. Fibers are often generated from textiles and fabrics, and their abundance is often higher in densely populated areas with textile industries and laundry near or in the water bodies. Indeed, a study by Sá et al. (2022) on the distribution of MPs in Lis River, Portugal, found a higher abundance of fibers and fragments near sites where manual washing of clothes was common, compared to sites with no laundry. Further, the breakdown of macroplastics such as PE and PP often creates irregularly shaped MPs, known as fragments or films (Browne et al. 2011). As such, the source and transportation medium of MPs largely determine their fate.

Abundance and distribution of MPs in riverine ecosystems

Research attention is being given to MP abundance and distribution in riverine ecosystems, with representative studies worldwide (Fig. 1; Tables 2 and 3). Out of the 183 reviewed studies, Asia had the highest number of reported studies ($n=85$), followed by Europe ($n=39$) and North America ($n=21$). The high number of studies could be ascribed to the advanced industrialization and high population (anthropogenic activities), which are the main drivers of MP pollution. These countries have invested in research and have developed institutional infrastructure compared to low-developed countries of Africa and Oceania. Indeed, the countries with high MP studies have already set policies to reduce MP pollution such as banning of single-use plastics and MPs in personal care products (Xanthos and Walker 2017; Kentin and Kaarto 2018; Hoang et al. 2022).

The abundance and distribution of MPs in riverine ecosystems vary, depending on geographical location, seasonality, and hydrological conditions (Hamid et al. 2018; Wu et al. 2020; Sá et al. 2022). In some riverine ecosystems, water levels and flow velocity are reduced during the dry season, increasing the residence time of MP particles and thus their higher abundance in upstream reaches (Nizzetto et al. 2016; Wicaksono et al. 2021; Sá et al. 2022). Meanwhile, precipitation during the wet season increases the water volume, leading to lower concentrations of MPs due to dilution (Wang et al. 2021a; Wicaksono et al. 2021). The higher flow velocity in the wet season increases runoff, washing MPs to downstream areas, as well as contributing to the fragmentation of larger plastic materials due to abrasion from sediment materials (Sá et al. 2022). Flooding is an influential factor determining the abundance, distribution, as well as characteristics of MPs. de Carvalho et al. (2021) found that the MP concentration in the Garonne River, France, increases by five- to eight-fold during flooding episodes, especially in the downstream reaches. Meanwhile, the concentration of larger particles increased on the water surface during flooding episodes, with the likely cause being the mobilization of plastic materials from the surrounding areas by increased water volume, alongside with high flow velocity (Cheung et al. 2019; de Carvalho et al. 2021).

Increased abundance and diversity (color, type, and polymer) of MPs are reported around and downstream urban centers, industrial parks, and wastewater treatment plants (de Carvalho et al. 2021). Certainly, the highest MP abundance in both water and sediments has been observed in highly urbanized and industrialized regions of Indonesia,

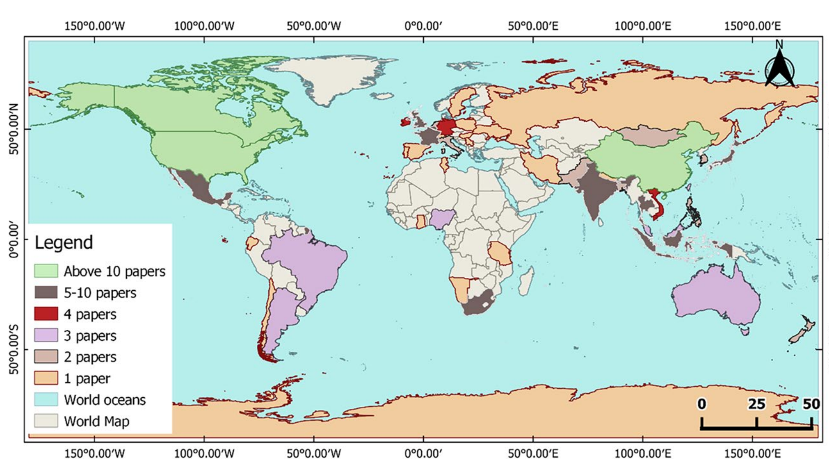

Fig. 1 Global distribution of studies on MP pollution in riverine ecosystems (based on 183 articles)

Table 2 Microplastic pollution reported in riverine water and sediments

Riverine system	Country	Organic matter digestion	Density separation	Size range (mm)	Observation	Abundance: mean (range)	Types	Polymers	Reference
Goulburn River catchment	Australia			0.02–	SM, FTIR	0.4 ± 0.3 items/L	Fiber, fragment, pellet	PES, PA, Acrylic, PE, elastine, PP	Nan et al. 2020
Ganges River	Bangladesh and India			0.3–5	SM, FTIR		Fiber, fragment	Acrylic, PET, PVC, PES, PA	Napper et al. 2021
Sinos River basin	Brazil	KOH			NR, FM	330.2 piece/L	Fiber, pellet, film		Ferraz et al. 2020
Ottawa River	Canada	H_2O_2			SM	1.35 pieces/m^3	Fiber, bead		Vermaire et al. 2017
North Saskatchewan River	Canada	Fe(II) + H_2O_2	$ZnCl_2$	0.053– > 1	SM, MR	26.2 ± 18.4 (4.6–88.3) particles/m^3	Fiber, fragment, film, sphere	PE, PES, PP, PU, PE-co-PP, PET, PVA, PVC, acrylic	Bujaczek et al. 2021
Biobío River catchment	Chile	NaClO	NaCl		SM, FTIR-DRIFTS, ATR-FTIR	22 ± 0.4 particles/m^3	Fragment, fiber	PVC, PE, PET, PS, PP, PMMA, PAGE	Correa-Araneda et al. 2022
Manas River basin	China	H_2O_2		< 0.1–5	FM, SEM, μ-FTIR	17 ± 4 items/L (dry), 14 ± 2 items/L (wet)	Fiber, fragment, film	PP, PET, PE, PS, PA, PVC	Wang et al. 2021a
Yellow River	China	H_2O_2		0.05–5	SM, SEM, ATR-FTIR	930 (dry season), 497 (wet) item/L	Fiber, fragment, particle	PE, PP, PS	Han et al. 2020
Fenghua River	China	H_2O_2	NaCl	0.5–5	SM, ATR-FTIR	(300–4000) n/m^3	Fiber, fragment, film, pellet/foam	PP, PE	Xu et al. 2021
Pearl River Delta	China	H_2O_2		< 0.1– > 0.5	SM, FTIR		Granular, strip, lump, rod-like, fragment, fiber	PA, PE, PS, PP, PET, PAN	Gao et al. 2022
Chishui River	China	H_2O_2		> 0.05–5	MM, ATR-FTIR	(1.8–14.3) items/L	Fiber, block, foam, film	PE, PP, PS, PVC	Li et al. 2021a
Minjiang River system	China	H_2O_2		0.20–5	SM, SEM, FTIR	8.8 (5–10.5) items/L	fragment, particle, fiber	PET, PA, EVOH, PE, PVC	Chen et al. 2022
Pearl River	China	H_2O_2		0.05–5	SM, MR	19,860 (8725–53,250) items/m^3	Film, granule, fiber	PA, CP, PP, PE, PVC	Yan et al. 2019

Table 2 (continued)

Riverine system	Country	Organic matter digestion	Density separation	Size range (mm)	Observation	Abundance: mean (range)	Types	Polymers	Reference
Pearl River (Xijiang River)	China	H_2O_2		0.01–5	SM, FTIR	7.67 ± 2.62 items/L	Fragment, fiber, film	POE, PET, PUA, EP, PF, silastic, PA, PE	Mai et al. 2021
Brahmaputra, Buqu, Naqu, Lhasa and Nyang Rivers	China	H_2O_2 + Fe(II)SO_4	$ZnCl_2$	< 0.5–5	SM, MR	(483–967) items/m^3	Fiber, fragment, pellet	PE, PET, PA, PP, PS	Jiang et al. 2019
Qing River	China	H_2O_2 + $FeSO_4$	$ZnCl_2$	0.001–5	SM, DM, FTIR	0.2 ± 0.1 (high flow), 0.3 ± 0.2 (low flow) particles/L	Fragment, fiber, film, pellet	EPR, PE, PU, PET, PS, PP, ER, MF, PVC, PS/PE, PE/PVA, PEA, acrylic, PS/PAC	Wang et al. 2021a
Qin River	China	H_2O_2	NaCl	0.025–5	SM, μFTIR		Fiber, fragment, sheet, foam, line	PE, PP, PET, PA, PVA, LDPE, PAN, CP, PS	Zhang et al. 2020
West River	China	H_2O_2		< 0.5 – 5	MM, FTIR	(3–9.9) items/L	Fiber, film, pellet, fragment	PP, PE, PS	Huang et al. 2021
Yongjiang River	China	Fe(II) + H_2O_2		0.05–5	SM, MR	2345 ± 1858 (500–7700) n/m^3	Fiber, fragment, film, pellet	PE, PET, PP, PA, PS	Zhang et al. 2019
Beijiang and Pearl Rivers	China	H_2O_2		0.01–5	SM	3183, Beijang; 7571, Pearl (400–18,200) items/m^3	Fragment, fiber, sphere, films		Wang et al. 2020b
Manas River Basin	China	H_2O_2	NaCl	< 0.1–5	FM, SEM, EDS, μ-FTIR	(21 ± 3–43 ± 8) items/L	Fiber, fragment, film	PP, PET, PA, PE, PS, PVC	Wang et al. 2020a
Tuojiang River basin	China			< 0.5–5	SM, SEM, FTIR	(911.6 ± 199.7–3395.3 ± 707.2) items/m^3	Fiber	PP	Zhou et al. 2020
Yulin River	China	H_2O_2		0.64–5	SM, MR	(0.01–0.6) items/L	Pellet/foam, line/fiber, film/fragment	PE, PP, PS	Mao et al. 2020

Table 2 (continued)

Riverine system	Country	Organic matter digestion	Density separation	Size range (mm)	Observation	Abundance: mean (range)	Types	Polymers	Reference
Pearl River (Guangzhou)	China	H_2O_2	NaCl	0.02–5	SM, µ-FTIR	2724 (379–7924) items/m^3	Fiber, fragment, film	PP, PE, PET	Lin et al. 2018
Wei River	China	H_2O_2	NaCl	< 0.5–5	MM SM, SEM	(3.7–10.7) items /L	Fiber, fragment, film, pellet, foam	PE, PVC, PS	Ding et al. 2019
Pearl River catchment	China	H_2O_2		0.16–5	SM, µ-FTIR	0.6 ± 0.7 items/L	Sheet, fragment, fiber, foam, spherule	PP, PE, PE-PP, PET, PVC, PA, PVA, EVA, Alkyd, ABS, PC, PAN, PU, SAN	Fan et al. 2019
Yangtze River (middle and lower reaches)	China			0.3–5	SM, MR	492,000 (195,000–492,000) items/km^2	Sheet, fragment, foam, line	PP, PE, PA, POM, EVA, PS	Xiong et al. 2019
Minjiang and Neijiang Rivers	China			0.089–4.72	SM	15.9 ± 3.1 (6.1–44.1) pieces/L	Fiber, fragment, particle	PP, PE, PA, PAN, PP-PE, ABS, PVC, PS, PU	Li et al. 2021b
Qinhuai River	China	Fenton's reagent		0.054–5	SM, MR	(1467 ± 223–20,567 ± 3233) items/m^3	Fragment, fiber, film, foam, pellet	PP, PE, PET, PS, PVC	Yan et al. 2021
Qiantang River	China	H_2O_2	NaCl		SM, FTIR	1183 ± 269 (50–6517) particles/m^3	Fiber, fragment, film, granule, foam	PA, PET, PE, PES, PP, PU, PS, PTFE	Zhao et al. 2020
Haihe River	China	H_2O_2	NaCl, centrifugation		SM, µFTIR	11.1 ± 4.4 (5.6–31.4) items/L	Fiber/line, fragment, film, pellet, foam	PE, PP, PS, PES, PET	Liu et al. 2021
Zhangjiang River	China	$FeSO_4 + H_2O_2$	NaCl	0.3–5	MR	246 (50–725) items/m^3	Fragment, fiber, line, pellets, film, foam	PP, PE, PE-PP, PES, PS, PET	Pan et al. 2020
Chin Ling-Wei River	China	H_2O_2		< 0.1–5	SM FTIR-ATR	9.8 (2.3–21.1) items/L	Fragment, film, fiber	PE, PP, PS, PA, PVC, PET	Bian et al. 2022

Table 2 (continued)

Riverine system	Country	Organic matter digestion	Density separation	Size range (mm)	Observation	Abundance: mean (range)	Types	Polymers	Reference
Suzhou and Huangpu Rivers	China	KOH		0.02–5	SM, ATR-µFTIR	(0.1–7.4) items/L	Fiber, fragment, pellet, film	PES, PP	Luo et al. 2019
Guayllabamba River basin	Ecuador			0.3–*x*	SM	(0.7–1584.2) MP/m^3	Fiber, film, fragment		Donoso and Rios-Touma 2020
Marne and Seine Rivers	France	*SDS, biozyme (F + SE),* H_2O_2	$ZnCl_2$	0.05–5	SM, FTIR	66.2 (38.2–101.6) $fibers/m^3$	Fibers	PET, PP, PA, PET-PU	Dris et al. 2018
Garonne River	France	KOH, H_2O_2		0.7–5	SM, ATR-FTIR	0.2 ± 0.5 (0–3.4) $particles/m^3$		PE, PP	de Carvalho et al. 2021
River Elbe	Germany	$KOH + H_2O_2$, formic acid	HCO_2K	0.15–5	SM, pyr-GCMS, ATR-FTIR	5.6 ± 4.3 (0.9–12.2) $particles/m^3$	Fiber, fragment, sphere, foil	PE, PP, PS	Scherer et al. 2020
Saynbach Stream	Germany				DM, µFTIR	0.003 ± 0.001 (0.002–0.007) MPs/mL	Fragment, fiber, film	UF, SIR	Ehlers et al. 2019
Confluence of Elbe and Mulde Rivers	Germany	H_2O_2		0.02–*x*	DM, pyr-GCMS		Fiber, sphere, film, fragment	PS, PE, PP	Laermanns et al. 2021
Maozhou River	Hong Kong	H_2O_2		0.001–5	ATR-µFTIR	(3.5 ± 1–25.5 ± 3.5) items/L	Fragment, foam, film, fiber	PE, PS, PP, PVC, PA, PET, PVA, PU, ABS	Wu et al. 2020
Lam Tsuen River	Hong Kong, China	H_2O_2		0.355–4.75	SM, ATR-FTIR	7.4 ± 3.7 $pieces/m^3$	Fiber, film, fragment, PS foam, pellet	PP/EPR, PP, PE, LDPE	Cheung et al. 2019
Kosasthalaiyar, Muthirappuzhayar and Adyar Rivers	India		Canola oil	0.335–5	FTIR	(0–1.82) particles/L	Fiber, film, fragment	PE, PP, PS	Lechthaler et al. 2021
Netravathi River	India	$H_2O_2 + Fe(II)$	$ZnCl_2$	0.3–5	SM, ATR-FTIR	288 (56–2,328) $pieces/m^3$	Fiber, film, foam, fragment, pellet	PE, PET, PP, PVC	Amrutha and Warrier 2020
Ganga River	India	$FeSO_4 + H_2O_2$	NaCl	0.3–7.5	SM, ATR-FTIR	466 (380–684) items/1000 m^3	Film, fragment, foam	PE, PVC, CP, PES, PTFE: propene	Singh et al. 2021

Table 2 (continued)

Riverine system	Country	Organic matter digestion	Density separation	Size range (mm)	Observation	Abundance: mean (range)	Types	Polymers	Reference
Tallo River	Indonesia	KOH		0.33–5	SM, FTIR	$(0.7 \pm 0.5–3.4 \pm 0.1)$ item/m^3	Fragments, fiber, film, pellets,	PE, PP, PBS	Wicaksono et al. 2021
Surabaya River	Indonesia	$Fe(II) + H_2O_2$	NaCl	0.333–5	SM, FTIR	(0.8–43.1) particles/m^3	Film, fragment, foam, fiber	LDPE, PP, PS, PET	Lestari et al. 2020
Ciwalengke River	Indonesia			0.05–2	SM	5.9 ± 3.3 particles/L	Fiber, fragment	PES, PA	Alam et al. 2019
Citarum River	Indonesia	H_2O_2	NaI	0.1–5	FTIR	(0–11) particles/20 mL	Fragment, fiber	PP, PE, PA, PET, PS, Alkyd	Jeong et al. 2021
Brantas River	Indonesia	H_2O_2		0.3–5	CM, FTIR	(133–5467) particles/m^3	Fragment, fiber, film, pellet	PE, PVC, PC, PA	Buwono et al. 2021
Krukut River	Indonesia	$Fe(II) + H_2O_2$	NaCl		SM	$(215 \pm 1.5–265 \pm 5)$ particle/100 mL	Fiber, pellet, fragment, film		Azizi et al. 2022
Mugnone Creek	Italy				SM, FPA-FTIR	(833–16,000) items/m^3	Fiber, fragment, film, pellet	PA, PET, PP + PE, PU, SBR, PTFE, PP	Rimondi et al. 2022
Ofanto river	Italy	$H_2O_2 + Fe(II)$	NaCl	< 1–5	DM, Py-GC–MS	13 ± 5 particles/m^3	Fragment, flake, line, fiber, pellet, foam	PE, PS, PP, PVC, PU	Campanale et al. 2020b
River Mignone	Italy	H_2O_2		1–5	SM, FTIR-ATR	0.3 (0.03–1.2) items/L	Fiber, fragment, film	PE	Gallitelli et al. 2020
Vipacco River	Italy				SM, µFTIR	3.7 ± 2.1 MPs/m^3/min	Fiber, sphere, fragment	PS, PET, PE, PA, PP, PU	Piccardo et al. 2021
Vipacco River	Italy				SM, µFTIR		Fiber, spherule, fragment	PS, PET, PE, PA, PES, PU, PP	Bertoli et al. 2022
Ohori River and Tone-Unga Canal	Japan	H_2O_2			SM, ATR-FTIR			PP, PE, PET, PE-PP	Tanaka et al. 2022
Agricultural and urban rivers (n-29)	Japan				SM, FTIR	1.6 ± 2.3 (0–12) pieces/m^3		PE, PP, PS, acryl, PA, EVA, polyacetylene	Kataoka et al. 2019

Table 2 (continued)

Riverine system	Country	Organic matter digestion	Density separation	Size range (mm)	Observation	Abundance: mean (range)	Types	Polymers	Reference
Tsurumi River	Japan	H_2O_2	NaI	0.02–5	µFTIR	(298 ± 105–1240 ± 295) particles/m^3		PE, PP, Alkyd, SBR, PVME, PVC, PVOH, PVA, PU, PS, PPS, PMMA, PIP, PET, PA, Epoxy, AS, ABS	Kameda et al. 2021
Awano, Ayaragi, Asa and Majime Rivers	Japan	$FeSO_4 + H_2O_2$	$ZnCl_2$	0.05–1	DM, ATR-FTIR	164.8 ± 171.9 n/L	Fiber, fragment, film	PP, PE, PET, PAN, Vinylon, PS, PPS, PVA, PA, ABS, EPDM, PCL, PBT	Kabir et al. 2021
Arakawa River watershed	Japan	H_2O_2		0.5–5	SM, ATR-FTIR	1.8 pieces/m^3	Fragment	PE, PP, PS	Sankoda and Yamada 2021
Ballenas Stream	Antarctica	H_2O_2	NaCl	0.1–5	SM, µFTIR	0.95 (0.47–1.43) items/1000 m^3	Fiber, film	PES, Acrylic, PTFE	González-Pleiter et al. 2020
Klang River	Malaysia	$FeSO_4 + H_2O_2$	NaCl	< 0.3–5	SM, ATR-FTIR	2.47 (0.5–4.5) particles/L	Fiber, fragment, pellet	PA, PE,	Zaki et al. 2021
Langat River	Malaysia	H_2O_2 + Fe(II)		0.53–5	SM, FTIR	4.4 ± 5.1 particles/L	Fiber, fragment, film, bead	PET, PE, PP, EVA, PVDC, PS	Chen et al. 2021
Selenga River	Mongolia	H_2O_2			DM, µFTIR	120.1 ± 121.5 items/m^2	Foam	PS, PE	Battulga et al. 2019
Khumbu and Imja Streams (Mt Everest)	Nepal			0.03–3.8	SM, FTIR	1 ± 0.3 (0–2 MP/L	Fiber	PES, acrylic, PA, PP	Napper et al. 2020
Koshi River	Nepal, China	H_2O_2		0.1–5	SM, ATR-FTIR	202 ± 100 (50 ± 11–325 ± 35) items/m^3	Fiber, pellet	PE, PP, PET, PS	Yang et al. 2021

Table 2 (continued)

Riverine system	Country	Organic matter digestion	Density separation	Size range (mm)	Observation	Abundance: mean (range)	Types	Polymers	Reference
Rhine River	The Netherlands, France, Germany, Switzerland	SDS, Biozym (F + SE), H_2O_2, chitinase, cellulase	NaCl	0.3–5	SM, FTIR	892,777 particles/km^2	Spherule, fiber, Fragment, foam, pellet, foils	PS, PP, acrylate, PES, PVC, PMMA,	Mani et al. 2015
Urban streams ($n = 52$)	New Zealand	H_2O_2 + Fe(II)	NaCl	0.25–5	SM	(1–44.8) items/m^3	Foam, fiber, fragment, beads		Mora-Teddy and Matthaei 2020
Auckland streams ($n = 18$)	New Zealand	H_2O_2, Fe(II) + H_2SO_4	NaCl	0.063–5	SM, FTIR	(17–303) items/m^3	Fragment, fiber, film, pellet, foam	PHM, EEAC, PE, PVC, PP, LDPE, EVA, EPDT	Dikareva and Simon 2019
Obiaraedu, Nwangele, Okumpi, Ogbajarajara, and Onuezuze Rivers	Nigeria				SM, HNT	(440–1556) particles/L	Fragment, fiber, film		Ebere et al. 2019
Swat River	Pakistan	H_2O_2	$ZnCl_2$	0.5–5	DM, FTIR-ATR		Fragment, fiber, pellet, film, foam	PE, PVC, PET, PS, PP	Khan et al. 2022
Słupia and Łupawa Rivers	Poland	$FeSO_4$ + H_2O_2		< 0.065–5	SM, ATR-FTIR		Fragment, granule, fiber	PE, PVC, PP, PES, PS	Piskuła and Astel 2022
Antuã River	Portugal	H_2O_2 + Fe(II)	$ZnCl_2$	0.055–5	SM, ATR-FTIR	(58–1265) items/m^3	Foam, fiber	PE, PP, PS, PET, PVA, EVA, PTFE, PMMA, PEA, SBR	Rodrigues et al. 2018
Lis River	Portugal	KOH		< 0.15–5	SM, ATR-FTIR	203.6 ± 727.8 (0.1–3422.2) items/m^3	Fragment, fiber, foam, pellet, bead, film	PE, PAcr, PS, PP, PVC, PA	Sá et al. 2022
Douro River	Portugal	H_2O_2 + Fe(II)		0.002–5	NR, SM	(< 1–334) MP/L	Fragment, sphere		Prata et al. 2021
Henares River catchment	Portugal	Fenton's reagent	NaI	0.055–5	DM, FTIR		Fragment, fiber, bead, granule, film, foam	PE, PP, PES, acrylic, PS, EPR, PVC	Schell et al. 2021

Table 2 (continued)

Riverine system	Country	Organic matter digestion	Density separation	Size range (mm)	Observation	Abundance: mean (range)	Types	Polymers	Reference
Ob and Tom Rivers	Siberia, Russia	Fe(II) + H_2O_2	NaCl		SM	(26.5 ± 11.8–114 ± 6.8) items/m^3	Fragment, fiber, film, sphere		Frank et al. 2021
Braamfontein Spruit Stream	South Africa	KOH		< 0.053–5	SM	705 (160–2080) particles/m^3	Filament, round, angular		Dahms et al. 2020
Vaal River	South Africa	$FeSO_4 + H_2O_2$	NaCl, NaI	< 0.5–5	SM, SEM, MR	0.6 ± 0.6 (0.1–2.5) particles/m^3	Fiber, fragment, film, pellet	HDPE, LDPE, PP	Ramaremisa et al. 2022
Han River and tributaries	South Korea	H_2O_2		0.1–5	FTIR	(0–234.5) particles/m^3	Fragment, fiber	silicone, PS, PP, PU, PES, PTFE, PE, PVC, acrylic, PA	Park et al. 2020
Nakdong River	South Korea	H_2O_2 + Fe(II)	LMT	0.02–5	ATR-FTIR	(293 ± 83–4760 ± 5242) particles/m^3	Fragment, fiber, sphere, film	PE, PES, PE, PA, PS, alkyd, acrylic, EVA, PU, PVC	Eo et al. 2019
Rhine River	Switzerland and Germany		Ethanol, castor oil	0.3–5	SM, ATR-FTIR	(0.04–9.97) MP/m^3	Fragment, spherule, film, line, pellet	PE, PP, PS, PC, PVC, PA, PES, PSF, EVA, acrylates/PU, ABS	Mani and Burkhardt-Holm 2020
Fengshan River system	Taiwan	H_2O_2	$ZnCl_2$	0.05–5	SM, ATR-FTIR	(334–1058) items/m^3	Fiber, fragment	PE, PET, PA, PES, PS, PU, PAC, PF, PP, PVC, EVA, PC	Tien et al. 2020
Tamsui River and tributaries	Taiwan	H_2O_2		0.3–5	SM, FTIR	(2.5 ± 1.8–83.7 ± 70.8) particles/m^3	Fragment, film, foam, pellet, bead, line		Wong et al. 2020
River Themi	Tanzania	Fe(II) + H_2O_2	$Na_2WO_4.2H_2O$	0.05–5	SM, ATR-FTIR		Fiber, fragment, film	PS, PE	Kundu et al. 2022
Tapi-Phumduang River system	Thailand			0.005–5	SM, FTIR	1.8 ± 0.3 (0.4 ± 0.1–2.8 ± 0.1) items/L	Fiber, fragment	PP, PE, PET, PA	Chinfak et al. 2021

Table 2 (continued)

Riverine system	Country	Organic matter digestion	Density separation	Size range (mm)	Observation	Abundance: mean (range)	Types	Polymers	Reference
Chao Phraya River	Thailand	$H_2O_2 + Fe(II)$	NaI	0.53–5	SM, FTIR	104 particles/m^3	Fragments, pellet, fiber, film	PP, PE, PS, PES, EVA, CP, PTFE	Ta et al. 2020
Saen Saeb Urban Canal	Thailand	$Fe(II) + H_2O_2$	NaCl	0.1–1	SM, FTIR	370 ± 140 MP/m^3	Film, fiber, fragment, pellet	PP, PE, PVS, PET, PVC, PS, PA	Eamrat et al. 2022
Chao Phraya River	Thailand	$Fe(II) + H_2O_2$	NaI, < 500 µm		CM, NR, µ-FTIR	80 ± 65 (48 ± 11–155 ± 16) items/m^3	Fragment, fiber, film, pellet	PP, PS, PE, CP, PU, PB	Ta and Babel 2020
Chao Phraya, Citarum and Saigon River	Thailand, Indonesia, Vietnam	$H_2O_2 + Fe(II)$	NaI	0.05–5	SM, ATR-FTIR	80 ± 60, 12 ± 6, 68 ± 20 items/m^3	Fragment, fiber, film, pellet	PP, PE, PP-PE, PES, PU, CP, PB, PS, PET	Babel et al. 2022
River Thames	UK			0.032–5	SM, µFTIR, ATR-FTIR		Film, fragment, bead, glitter, nurdle	PE, PP, PES, PP-PE, TPU, Acrylic, PA	Rowley et al. 2020
Thames River (New Year period effect)	UK				DM, FTIR-ATR	51.2 (22–510) pieces/L	Fibers, fragment, glitter	PCP, PVC, PET, CPE, HDPE, PTFE, PMA, ABS, PES, PP, PS, PVA, PVC	Devereux et al. 2022
Clackamas River and Johnson Creek	USA	KOH	Saline solution	0.063–5	SM, µ-FTIR		Fragment, fiber, film, foam	PE, PP, PET, PS, CP, EVA, PVA, SBR	Talbot et al. 2022
Streams in Chicago Metropolitan Area ($n = 10$)	USA	$Fe(II) + H_2O_2$	NaCl	0.33–0.48	SM, pyr-GCMS	1,338,757 (15,520–4,721,709) pieces/day	Pellet, fragment, fiber, Film, foam	PE, PP, PS, EPDM,	McCormick et al. 2016
Gallatin River and tributaries	USA			0.1–5	SM, µFTIR	1.2 (0–67.5) particles/L	Fiber, fragment, beads	PES, PET, BR, Acrylic, PA, PP, urethane, PVOH, AN, PVDF, NP	Barrows et al. 2018

Table 2 (continued)

Riverine system	Country	Organic matter digestion	Density separation	Size range (mm)	Observation	Abundance: mean (range)	Types	Polymers	Reference
The North Shore Channel	USA	$Fe(II) + H_2O_2$	NaCl	0.333–2	SM	1.9 ± 0.8–17.9 ± 11.1 no./m^3	Fragment, fiber, pellet, styrofoam		McCormick et al. 2014
Mohawk River	USA	WPO			SM	(0.1–118.7) particles/m^3	Fiber, fragment, foam, film, pellet/bead		Smith et al. 2019
Raritan River	USA	$Fe(II) + H_2O_2$	NaCl	0.125–2	SM	(24 ± 11.4 (upstream WWTP), 71.7 ± 60.2 (downstream)) no/m^3	Secondary, primary		Estahbanati and Fahrenfeld 2016
White River	USA	WPO	NaCl	0.25–0.5	SM, HNT	0.71 items/m^3	Fiber, fragment, foam, pellets		Hylton et al. 2018
Raritan and Passaic River	USA	$FeSO_4 + H_2O_2$			SM, Pyr-GCMS	(~28,000–> 3,000,000) particles/km^2	Fragment, foam, line, film		Ravit et al. 2017
Saigon River and Ho Chi Minh City Canals	Vietnam	$H_2O_2 + Fe(II)$	$NaCl + ZnCl_2$	0.5–5	DM, ATR-FTIR	0.6 ± 0.4, river; 104.2 ± 162.4, canals (0.2–666.7) pieces/m^3	Granule/pellet, fragment, fiber	PP, PE, EVA, PP-PE, PET, ABS, PE-ABS, PP-EVA, PA, PTFE, PMMA	Nguyen et al. 2022
Saigon urban canals	Vietnam	H_2O_2			SM, MR	38.5 ± 24.9 (30–250) MP/L	Fiber, fragment, film, foam, pellet/granule	PA, PET, PE, PP, PVC, PS, PMMA	Khuyen et al. 2022
Saigon River	Vietnam	SDS, biozym (SE + F), H_2O_2	$ZnCl_2$	0.05–4.85	SM, FTIR	(172,000–519,000) items/m^3, fibers,	Fiber, fragment	PES, PE, PP, PE-PP, acrylic, PVC, PS, PA	Lahens et al. 2018
Sediments									
Paraná River	Argentina	H_2O_2	NaCl	0.35–5	SM, FTIR	(75–34,443) MP/m^2	Film, fiber, styrofoam		Blettler et al. 2019
Brisbane River	Australia		$ZnCl_2$	< 1–5	SM, ATR-FTIR	(10–520) items/kg	Film, fragment, fiber	PE, PA, PP, PET	He et al. 2020
Scheldt River	Belgium	H_2O_2	NaI	0.015–1	DM, MR	(1840 ± 2407–$63{,}112 \pm 24{,}628$) MP/kg	Fragment, bead	PP, PET, PE, PVC, PS, Teflon, PA	De Troyer 2015

Table 2 (continued)

Riverine system	Country	Organic matter digestion	Density separation	Size range (mm)	Observation	Abundance: mean (range)	Types	Polymers	Reference
Solimões, Negro, and Amazon rivers	Brazil	H_2O_2	$ZnCl_2$	0.063–5	SM	(417–5725, 0–5725) particle/kg			Gerolin et al. 2020
St. Lawrence River	Canada			< 0.5–*x*	SM, DSC	13,832 ± 13,677 microbeads/m^2			Castañeda et al. 2014
Ottawa River	Canada	H_2O_2	NaCl		SM	0.22 pieces/g	Fiber		Vermaire et al. 2017
St. Lawrence River	Canada	H_2O_2	Canola oil	0.001–5	FM	832 (65–7,562) plastics/kg	Bead, fragment, fiber		Crew et al. 2020
Thames River	Canada		SPT solution		SM, ATR-μFTIR	(6–2,444) particles/kg	Fiber, fragment, bead	PET, PE, Alkyd, PA, PAN, CSPE, PE-PP, PU, PVC, EP, VA, NC, Acrylic/polyacrylic,	Corcoran et al. 2020
Yangtze River and tributaries	China		NaCl, $ZnCl_2$	< 0.5–5	SM, ATR-FTIR	(35–51,968) particles/kg	Flake, film, foam, fiber, fragment, sponge, pellet	PP, PS, PE	Li et al. 2022
Liangfeng River	China	H_2O_2	NaCl	0.05–5	LCM, FTIR	33,200 ± 11,990 (6950–149,350) items/kg	Fiber, fragment, film	PE, PP, PVC, PS, PA	Xia et al. 2021
Buqu, Naqu, Lhasa, Brahmaputra and Nyang Rivers	China	$H_2O_2 + FeSO_4$	$ZnCl_2$	< 0.5–5	SM, MR	(50–195) items/kg	Fiber, fragment, pellet	PET, PA, PE, PS, PP	Jiang et al. 2019
Beijiang River	China		NaCl		DM, SEM, EDS, ICP-MS, μ-FTIR	(178 ± 69–544 ± 107) items/kg		PE, PP	Wang et al. 2017
Shuangtaizi and Daliao River	China	H_2O_2	$ZnCl_2$		SM, DM, ATR-FTIR	(67–467) particles/kg	Film, fragment, fiber, pellet	PE, EPR, PP, PET, CPA, SBS, EVA	Xu et al. 2020
Qin river	China	H_2O_2	NaCl	0.3–5	SM, μFTIR		Fiber, fragment	PE, PP, PET, EVA	Zhang et al. 2020

Table 2 (continued)

Riverine system	Country	Organic matter digestion	Density separation	Size range (mm)	Observation	Abundance: mean (range)	Types	Polymers	Reference
West River	China	H_2O_2	Saturated salt	< 0.5–5	MM	(2560–10,240) items/kg	Fiber, film, fragment, pellet	PP, PE, PS, PVC, PET	Huang et al. 2021
Shanghai rivers (n = 6)	China		NaCl	< 0.1–5	SM, ATR-μFTIR	802 ± 594 items/kg	Sphere, fiber, fragment	PP, PES, phenoxy, PVS, rayon + PES	Peng et al. 2018
Yongjiang River	China	Fe(II) + H_2O_2	NaCl, NaI + KI	0.05–5	SM, Raman	285 ± 110 (90–550) pieces/kg	Fiber, fragment, film, pellet	PE, PP, PET, PA, PS	Zhang et al. 2019
River Yongfeng	China	H_2O_2 + HNO_3	NaCl	< 0.2–5	SM, EDS, FESEM, ATR-FTIR	26 ± 23 (5–72) items/kg	Film, line, fragment	PE, PP, PET, PS	Rao et al. 2020
Pearl River (Guangzhou)	China	KOH	NaCl	0.02–5	SM, μ-FTIR	1669 (80–9597) items/kg	Fiber, fragment, film	PE, PP	Lin et al. 2018
Wei River	China	H_2O_2	Saturated salt solution	< 0.5–5		(360–1320) items/kg	Fiber, film, fragment, pellet, foam	PE, PVC, PS	Ding et al. 2019
Pearl River catchment	China		HCO_2K	0.16–5	SM, μ-FTIR	685 ± 342 items/kg	Sheet, fiber, fragment, spherule, foam	PP, PE, PE-PP, PET, PVC, PVA, PA, EVA, Alkyd, ABS, PC, PU, PMMA, SAN	Fan et al. 2019
Yangtze River (middle & lower reaches)	China			0.3–5	SM, Raman	34 (7–66) items/kg		PP, PE, PS, PA, EVA	Xiong et al. 2019
Minjiang and Neijiang Rivers	China		Centrifugation	0.66–4.77	DM, FTIR	1391.3 ± 194.3 (573.8–2879) n/kg	Fragment, fiber, particle	PE, PP, PVC, PA, PS, PET, PAN, PU, PP-PE, ABS, PTFE	Li et al. 2021b
Qinhuai River	China	Fenton's reagent	NaCl, $ZnCl_2$	0.054–5	SM, μRaman	(1115 ± 85–6380 ± 280) items/kg	Fiber, fragment, foam, pellet, film	PP, PS, PE, PET	Yan et al. 2021
Qiantang River	China	KOH	$CaCl_2$	0.05–5	SM, μFTIR	0.2 ± 0.1 particles/g	Fragment, fiber	PE, PET, PS, PS-PP	Fraser et al. 2020

Table 2 (continued)

Riverine system	Country	Organic matter digestion	Density separation	Size range (mm)	Observation	Abundance: mean (range)	Types	Polymers	Reference
Haihe River	China	H_2O_2	NaCl		SM, μFTIR	4328 ± 2037 (2142–10,035) items/kg	Fiber/line, fragment, film, pellet	PE, PP, SBS, PVC, PMA, PES	Liu et al. 2021
Guayllabamba River basin	Ecuador	H_2O_2 + Fe(II)	NaCl		SM	(14.3–186.5) MP/kg	Fiber, film, fragment		Donoso and Rios-Touma 2020
Rhine and Main Rivers	Germany	$H_2O_2 + H_2SO_4$	NaCl	0.063–5	SM, ATR-FTIR	(228–3763) particles/kg	Fragments, fiber, film, pellets,	PS, PE, PP, PET, PVC, EPDM, PA	Klein et al. 2015
Roter Main River	Germany	enzymatic purification		0.02–5	ATR-FTIR	(0–0.0022 [0.5–5 mm]; ~ 10,000– > 50,000 [0.02–0.5 mm]) pieces/kg	Fragment, fiber	PAN, PS, epoxide, PTFE, PP, PU, PE	Frei et al. 2019
River Elbe	Germany	$H_2O_2 + H_2SO_4$	$ZnCl_2$	0.125–5	SM, pyr-GCMS, ATR-FTIR	3,350,000 ± 6,660,000 (22,600–22,700,000) piece/m^3	Sphere, fragment, fiber, foil	PE, PS, PP, ABS, PA, PET, PMMA	Scherer et al. 2020
Saynbach Stream	Germany	KOH + H_2O_2	HCO_2K		DM, μFTIR	0.09 ± 0.05 (0–0.26) MP/g	Fiber, film	PE, acryl, PES	Ehlers et al. 2019
Confluence of Elbe and Mulde Rivers	Germany	H_2O_2	HCO_2K		DM, py-GCMS		Sphere, fragment, film, fiber	PE, PS, PP	Laermanns et al. 2021
Maozhou River	Hong Kong	H_2O_2	$ZnCl_2$	0.001–5	ATR-μFTIR	(25 ± 5–560 ± 7) items/kg	Fragment, foam, fiber, film	PE, PVC, PS PP, PA, PET, PVA, PU, ABS	Wu et al. 2020
Kavery River	India	$FeSO_4 + H_2O_2$		< 0.1–5	SEM, EDS, ATR-FTIR	386.1 ± 84.5 (187 ± 103–699 ± 66) items/kg	Fragment, film, fiber, foam	PA, PP, PVC, PS, PEG, PET	Maheswaran et al. 2022
Netravathi River	India		$ZnCl_2$	0.3–5	SM, ATR-FTIR	96 (9.4–253.3) pieces/kg	Fragment, fiber, film, foam, pellet	PE, PET, PP	Amrutha and Warrier 2020
Ganga River	India		NaCl	0.3–7.5	SM, ATR-FTIR	25 (17–36) items/kg	Film, foam, filament, fragment	PE, PS, CP, PP, PE-PP PVT: butadiene, PVC, PBAN, PES	Singh et al. 2021
Brahmaputra and Indus Rivers	India	H_2O	$Na_2WO_4{\cdot}2H_2O$	0.02–5	FTIR, SEM, μFTIR	(20–340 (0.15–5 mm), 525–3485 (0.02–0.15 mm)) MP/kg	Fragment, fiber, bead	PP, PE, PA, PET, PTFE, PVC, PS	Tsering et al. 2021

Table 2 (continued)

Riverine system	Country	Organic matter digestion	Density separation	Size range (mm)	Observation	Abundance: mean (range)	Types	Polymers	Reference
Tallo River	Indonesia		NaCl	0.0045–5	SM	(16.7 ± 20.8–150 ± 36.1) items/kg	Fragments, fiber, film, pellets,	PE, PP, PES	Wicaksono et al. 2021
Ciwalengke River	Indonesia		NaCl	0.05–2	SM, Raman	3 ± 1.6 particles/100 g dw	Fiber, fragment	PES, PA,	Alam et al. 2019
Winongo, Code and Gadjahwong Rivers	Indonesia		NaCl	0.001–5	SM	635.8 ± 161.7 (279.3–1026.9) pieces/kg	Fragment, fiber, pellet, film		Utami et al. 2021a
Krukut River	Indonesia		NaCl		SM	(112 ± 3.5–150 ± 5) particle/kg	Fiber, pellet, fragment, film		Azizi et al. 2022
Progo River	Indonesia		NaCl	0.001–5	CM, FTIR	467.5 ± 225.9 (209.4–1173.3) pieces/kg	Fiber, fragment, film, pellet	PES	Utami et al. 2021b
River Barrow	Ireland		$ZnCl_2$	0.1–5	SM, ATR-FTIR	172.5 ± 116 MP/kg ww	Fiber, fragment	PP, PA, PET	Murphy et al. 2022
Mugnone Creek	Italy	H_2O_2	NaCl	*x*–2	SM, FPA-FTIR	860 ± 360 (500–1540) items/kg	Fiber, fragment, film		Rimondi et al. 2022
River Mignone	Italy			1–5	SM		Fiber, fragment, film	PE, PP, PA, PES	Gallitelli et al. 2020
Vipacco River	Italy		NaCl		SM, μFTIR	3.3 ± 4.2 MP/dm^3	Spherule, fiber, fragment	PS, PP, PA, PU	Piccardo et al. 2021
Vipacco River	Italy		NaCl	0.006–*x*	SM, μFTIR		Spherule, fiber, fragment	PS, PP, PES, PA, PU	Bertoli et al. 2022
Skudai and Tebrau Rivers	Malaysia	H_2O_2	NaCl	0.0012–5	SM	200 ± 80 (Skudai); 680 ± 140 (Tebrau) particles/kg	Film		Sarijan et al. 2018
Atoyac River basin	Mexico	H_2O_2	$ZnCl_2$		SM, SEM, SEM/EDS		Film, fragment, fiber, pellet		Shruti et al. 2019
Tuul River	Mongolia	Fe(II) + H_2O_2		0.28–3.41	DM, μFTIR	603 ± 251 (312–998) items/kg	Fiber, fragment, foam, film	PES, PA, PE, PS, PVC, ABS	Battulga et al. 2020
Iishana System	Namibia		NaCl	0.3–5	NR, SM, FTIR-ATR	(0–66) particle/kg	Fragment, film, fiber, pellet	PE, PP, PS, PA, PMMA	Faulstich et al. 2022

Table 2 (continued)

Riverine system	Country	Organic matter digestion	Density separation	Size range (mm)	Observation	Abundance: mean (range)	Types	Polymers	Reference
Koshi River	Nepal, China	$H_2O_2 + Fe(II)$	$ZnCl_2$	0.1–5	SM, ATR-μFTIR	58 ± 27 (15 ± 3–120 ± 23) items/kg,	Fiber, fragment	PE, PA, PET, PS	Yang et al. 2021
Dommel River	The Netherlands	H_2O_2	$ZnCl_2$	0.02–5	SM, FTIR-ATR, FTIR	(2910–16,740) particles/kg	Particle, fiber	APV, PCP, PE, PP, CPE, PS, PA, PC, EVA	Pan et al. 2021
Auckland streams ($n = 18$)	New Zealand		NaI	0.063–5	SM, LDPA, FTIR	(9–80) items/kg	Fragment, fiber, film, foam	PHM, EEAC, PE, PVC, PP, LDPE, EVA, EPDT	Dikareva and Simon 2019
Swat River	Pakistan	H_2O_2	$ZnCl_2$	0.5–5	DM, FTIR-ATR		Fragment, fiber, pellet, film	PE, PVC, PET, PP, PS	Khan et al. 2022
Antuã River	Portugal		$ZnCl_2$, SPP	0.055–5	SM	(18–629) items/kg	Fragment	PE, PP, PS, PET, PVA, EVA, PTFE, PMMA, PEA, SBR	Rodrigues et al. 2018
Lis River	Portugal	H_2O_2	$ZnCl_2$	0.063–5	SM		Fiber, fragments, pellets, films, beads	PET, PAcr, PS, PVC, PA	Sá et al. 2022
Henares River catchment	Portugal	Fenton's reagent	NaI	0.055–5	DM, FTIR	(49.7–2630) MP/kg	Fragment, fiber, glitter	PE, PP, PES, acrylic, PS, EPR, PVC	Schell et al. 2021
Braamfontein Spruit Stream	South Africa		NaCl	< 0.053–5	SM	166.8 (4–1347.5) particles/kg	Filament, angular		Dahms et al. 2020
Bloukrans River system	South Africa	hyper-saline solution		0.063–5	SM	(13.3–563.8) particles/kg			Nel et al. 2018
Vaal River	South Africa	KOH	NaI	< 0.5–5	SM, SEM, Raman	460 ± 280 (29–1100) particles/kg	Fragment, fiber, film	HDPE, LDPE, PP, PEVA, PU, PES	Ramaremisa et al. 2022
Mvudi River	South Africa		NaCl	0.063–5	SM	(13.0 ± 4.0–285.5 ± 44.5) MP/kg	Fiber, bead		Dalu et al. 2021
Vaal River	South Africa	KOH	NaI	< 0.5–5	SM, SEM, Raman	463.3 ± 284.1 (29.1–1095.9) particles/kg	Fragment, fiber, pellet	PE, PP, PEVA, PES, PU	Saad et al. 2022
Nakdong River	South Korea		LMT	0.02–5	ATR-FTIR	1970 ± 62 particles/kg	Fragment, fiber, sphere	PP, PE, PES, PVC, PS, acrylic, PU	Eo et al. 2019

Table 2 (continued)

Riverine system	Country	Organic matter digestion	Density separation	Size range (mm)	Observation	Abundance: mean (range)	Types	Polymers	Reference
Fengshan River system	Taiwan	H_2O_2	$ZnCl_2$	0.05–5	SM, ATR-FTIR	(508–3987) items/kg	Fiber, fragment	EP, PET, PF, PVA, PE, PA, PS, PES, PU, PP, PVC, EVA, PTFE	Tien et al. 2020
River Themi	Tanzania	Fe(II) + H_2O_2	$Na_2WO_4.2H_2O$	0.05–5	SM, ATR-FTIR		Fiber, fragment, film, bead	PE	Kundu et al. 2022
Tapi-Phumduang River system	Thailand		NaCl	0.005–5	SM, FTIR	79 ± 18.3 (55–160) items/kg	Fiber, fragment	PP, PE, PET, PA	Chinfak et al. 2021
Chao Phraya River	Thailand	H_2O_2 + Fe(II)	NaI	0.053–5	SM, FTIR	2290 particles/kg		PE, PP, PS, PES	Ta et al. 2020
lower Chao Phraya	Thailand	Fe(II) + H_2O_2 − < 500 µm	NaI		CM, NR, µ-FTIR	91 ± 13 items/kg	Fiber, fragment, film, pellet	PP, PE, PS, PES	Ta and Babel 2020
Streams in Bizerte Region (n = 7)	Tunisia		NaCl	0.2–5	SM, FTIR-ATR	(2340 ± 227.1–6920 ± 396) items/kg	Fiber, fragment, film	PP, PE	Toumi et al. 2019
River Tame	UK		$ZnCl_2$	0.063–4	SM, ATR-FTIR	165 particles/kg	Fragment, fiber, sphere, foams, films	PE, PVC, PMMA	Tibbetts et al. 2018
River Irwell	UK		NaCl		SM, ATR-FTIR	914 ± 844 (56–2543) particles/kg	Fragment, beads, fiber	PS, PE, PP, PET, PVA, acrylic/PAN, PES	Hurley et al. 2017
River Thames tributaries	UK		$ZnCl_2$	1–4	SM, Raman	(18.5 ± 4.2–66 ± 7.7) particles/100 g	Fragment, fiber, film	PET, PP, PASF PE, PS, PVC, PC, AN/PMMA	Horton et al. 2017
River Kelvin	UK		NaCl	0.011–2.8	SM, SEM/EDS	(161–432) MP/kg	Fiber, fragment, pellet		Blair et al. 2019
Bourne Stream	UK		$ZnCl_2$	0.038–5	SM, FTIR-ATR		Fragment, bead, fiber	POE, PS, PES, PA	Parker et al. 2022b
Tisza River	Ukraine and Hungary	H_2O_2	$ZnCl_2$	0.09–2	DM	3177 ± 1970 items/kg	Fiber, fragment		Kiss et al. 2021

Table 2 (continued)

Riverine system	Country	Organic matter digestion	Density separation	Size range (mm)	Observation	Abundance: mean (range)	Types	Polymers	Reference
Stroubles Creek, Roanoke River, James River	USA		NaCl		SM, ATR-FTIR	63 (17–180) particles/kg	Fragment, sphere, fiber	PP, PS, PET, PE	Christensen et al. 2020

Density separation: *SPP*, sodium polyphosphate; *LMT*, lithium metatungstate; *SPT*, sodium polytungstate. Observation: *SM*, binocular/stereo/light microscope; *DM*, dissecting microscope; *LCM*, laser confocal microscope; *FM*, fluorescent microscope; *MM*, metallographic microscope; *ICP-MS*, inductively coupled plasma mass spectrometry; *LDPA*, laser diffraction particle analyzer; *EDS*, energy dispersive X-ray spectroscopy; *Pyr-GCMS*, pyrolysis gas chromatography mass spectrometry; *(μ)FTIR*, (micro) Fourier transform infrared spectroscopy; *MR*, micro-Raman; *ATR*, attenuated total reflectance; *FPA-FTIR*, focal plane array FTIR; *DSC*, differential scanning calorimetry; *FESEM*, field emission scanning electron microscopy; *HNT*, hot needle test; *NR*, Nile Red solution. Polymers: *PET*, polyethylene terephthalate; *PP*, polypropylene; *PA*, polyamide (nylon); *PU*, polyurethane; *PE*, polyethylene; *PS*, polystyrene; *PVA*, polyvinyl acetate; *EVA*, ethylene–vinyl acetate; *PTFE*, polytetrafluorethylene; *PMMA*, polymethyl methacrylate; *PEA*, poly(ethyl acrylate); *SBR*, styrene butadiene rubber; *PVC*, polyvinyl chloride; *PAcr*, polyacrylate; *PSB*, poly(styrene-butadiene); *PES*, polyester; *EPDM*, ethylene-propylene diene rubber; *LDPE*, low density polyethylene; *ABS*, acrylonitrile butadiene styrene; *CP*, cellophane; *PAN*, polyacrylonitrile; *PAGE*, polyacrylamide; *EPR*, poly(ethylene-propylene); *PB*, polybutadiene; *PEG*, polyethylene glycol; *EVOH*, ethylene vinyl alcohol; *PBT*, polybutylene terephthalate; *PC*, polycarbonate; *BR*, butadiene/butyl rubber; *AN*, acrylonitrile; *PVDF*, poly(vinylidene fluoride); *PVOH*, polyvinyl alcohol; *NP*, neoprene; *POE*, polyolefin elastomer; *PUA*, polyurethane acrylate; *EP*, polyurethane acrylate; *PF*, phenol formaldehyde resin; *PS-DVB*, polystyrene-divinylbenzene; *PHM*, poly(hexadecyl) methacrylate; *EEAC*, propylene glycol monoleate; *EPDT*, ethylene propylene diene terpolymer; *PVME*, poly(vinyl methyl ether); *PPS*, poly(phenylene sulfide); *PIP*, polyisoprene; *CPA*, chlorinated polyalkene; *SBS*, poly(styrene–butadiene–styrene); *MF*, melamine formaldehyde; *PAC*, polyacrylate; *PVS*, poly(vinyl stearate); *PSF*, polysulfone; *UF*, urea formaldehyde resin; *PASF*, polyarylsulfone; *PVT*, polyvinyl toluene; *PBAN*, polybutadiene acrylonitrile; *PVDC*, polyvinylidene chloride; *VA*, vinyl acetate; *CSPE*, chlorosulfonated PE; *NC*, nitrocellulose; *SAN*, styrene acrylonitrile; *POM*, polyoxymethylene; *TPU*, thermoplastic rubber; *CPE*, chlorinated polyethylene; *APV*, acrylates/polyurethanes/varnish cluster; *PMA*, polymethyl acrylate; *SIR*, silicon rubber; *PCP*, polychloroprene; *PEVA*, polyethylene vinyl acetate

Nigeria, and China (Table 2). Up to 2,650,000 particles/m^3 were observed in the Krukut River in Jakarta, Indonesia (Azizi et al. 2022), while 1,566,000 particles/m^3 were found in the Nwangale River, Imo State, Nigeria (Ebere et al. 2019). Both Krukut and Nwangale rivers pass through urban and suburban areas, receiving domestic, municipal, industrial, and agricultural wastes, resulting from poor waste management practices (Ebere et al. 2019; Azizi et al. 2022). Mass gatherings (social events) around river systems lead to a sudden increase in MP pollution. Evidently, in Thames River, London (UK), MP abundance suddenly increased from 44.2 (44,200 particles/m^3) on December 31, 2019 to 510 particles/L (510,000 particles/m^3) on January 1, 2020 after New Year's celebration (Devereux et al. 2022), which can be attributed to plastic litter, including fireworks during the event.

Wastewater treatment plants (WWTPs) have been reported as major MP sources in riverine environments (Piskuła and Astel 2022). The observed MPs in the influents and effluents of WWTP are mainly fibers, fragments, and beads released into the system from laundry (household and commercial) and personal care products (Edo et al. 2020; Habib et al. 2022; Hoang et al. 2022). Indeed, the highest MP abundance in sediments in this review was 149,350 MP items/kg in Liangfeng River, China, downstream of a WWTP (Xia et al. 2021). In the Yangtze river, up to 51,968 items/kg were reported in sediment samples collected from Wuhan, a highly populated region in China (Li et al. 2022). Similar to surface water, high abundances of MPs in sediments are reported in urban and suburban areas with increased anthropogenic sources, such as industries, agriculture, and residential homes.

Certainly, MP pollution has been found to have a significant positive correlation with economic development and population density in the riverine catchment (Chen et al. 2022; Li et al. 2022). Urbanized areas with direct waste input in aquatic ecosystems tend to create local MP "hotspots," which can explain the higher MP abundance in the highly populated upstream reaches of the Tisza River than in downstream reaches (Kiss et al. 2021). Indeed, population density and human activities in the catchment are key factors in the spatial distribution of MPs in rivers (de Carvalho et al. 2021; Sá et al. 2022). Thus, the higher abundance of MPs reported downstream of urban and industrial centers is an indication of the interactive effect of urbanization (including population density) and flooding on MP pollution (Wu et al. 2020; de Carvalho et al. 2021). Human activities, such as washing of clothes within the rivers, directly release MPs in water, through wear and tear of fabric (Ebere et al. 2019). A study by Browne et al. (2011) showed that over 1900 fiber particles can be produced from a single garment per wash.

Generally, the size of MPs tends to reduce toward the river mouth, an indication of continued physical and

Table 3 Microplastic ingestion/uptake reported in riverine macroinvertebrates

Riverine system	Country	Macroinvertebrate studied	Order	Extraction	Observation	Abundance mean (range)	Types	Colors	Polymers	Reference
Goulburn River catchment	Australia	*Paratya australiensis*	Decapoda	NaOH	SM; FTIR	24 ± 31 items/g	Fiber, fragment	Blue, red, transparent	PES, PES + PA, PP + PE, PP, acrylic; PA	Nan et al. 2020
Grand River watershed	Canada	*Lasmigona costata*	Unionida	protease enzyme	SM; Raman Spectroscopy	0–7 piece/ind	Fragment, fiber		PP-co-PE, PP, P4VP, PEI, PAA, PDMS, PP-co-1-butene	Wardlaw and Prosser 2020
Ottawa River	Canada	Amphipoda, Gastropoda, Plecoptera, Decapoda, Bivalvia		H_2O_2	SM	0–101 MP/ind				D'Addario 2020
Rhine River	Germany	*Theodoxus fluviatilis*	Cycloneritida	$KOH + H_2O_2$	DM; μ-FTIR		Fiber	Black, white, blue	PA, PP	Akindele et al. 2019
Saynbach Stream	Germany	*Lepidostoma basale* (cases)	Trichoptera	H_2O_2; HCO_2K	DM; μ-FTIR	1.14 ± 0.28 MP/larval case	Film, fiber, fragment sphere	Blue, green	PP, PA, ABS, PE, PAM, TPU, PA-ABS, PVC, VE, PES	Ehlers et al. 2019
Treja and Mignone Rivers	Italy	*Simulium equinum; Simulium ornatum*	Diptera	H_2O_2	μ-FTIR	462 ± 30; 1101 ± 47 SMP/organism			PA, PO, PPA, HDPE, PFA, PP, PES, PEA, EPM, ECTFE, PEAA, PC/ABS, PA, BR, PBA, aramid, EVOH, PARA, FKM, PTFE,	Corami et al. 2022
River Mignone	Italy	Bereidae and Hydroptilidae cases	Trichoptera	H_2O_2	SM; FTIR-ATR	6.4 (1–16) piece/case	Fiber			Gallitelli et al. 2020
River Licenza	Italy	*Odontocerum albicorne (cases)*	Trichoptera	H_2O_2	SM; FTIR	17 lMP/case	Fiber, film fragment	Blue, black, red, white	PES, PA, PE, PP	Gallitelli et al. 2021

Table 3 (continued)

Riverine system	Country	Macroinvertebrate studied	Order	Extraction	Observation	Abundance mean (range)	Types	Colors	Polymers	Reference
Vipacco River	Italy	Hydracarina; Chironomidae; *Lymnaea; Asellus*; *Limnius, Elmis*, *Oulimnius, Stenelmis*; *Caenis*, *Potamanthus, Ephemera*; Gomphidae, Coenagrionidae, *Calopteryx*; *Leuctra; Hydropsyche*	Trombidiformes; Diptera; Basommatophora; Isopoda; Coleoptera; Ephemeroptera; Odonata; Plecoptera; Trichoptera	Creon enzyme	SM; μFTIR		Fiber	Blue, pink, green, black, grey, white, orange	PES	Bertoli et al. 2022
Dommel River	The Netherlands	Asellidae; Chironomidae; Astacidae; Gammaridae; Ephemeroptera; Tubificidae; Siallidae; Erpobdellidae; Calopterygidae, Coenagrionidae	Isopoda; Diptera; Decapoda; Amphipoda; Ephemeroptera; Tubificida; Megaloptera; Arhynchobdellida; Odonata	H_2O_2 + chitinase and protease enzymes	SM; FTIR-ATR	0–195 piece/ind	Particle, fiber		EPDM, PEC	Pan et al. 2021
Osun River system	Nigeria	*Lanistes varicus; Melanoides tuberculata*	Architaenioglossa; Neotaenioglossa	KOH + H_2O_2	DM; μ-FTIR		Fiber, film	Black, white, blue	PE	Akindele et al. 2019
Ogun and Osun Rivers	Nigeria	*Chironomus* sp.; *Siphlonurus* sp.; *Lestes viridis*	Diptera; Ephemeroptera; Odonata	KOH + H_2O_2	DM; μ-FTIR	291.8 ± 26.7; 62.4 ± 3.5; 43.3 ± 43.4 piece/g ww	Fiber, fragment		SEBS, ABS, CPE, PP, PES	Akindele et al. 2020

Table 3 (continued)

Riverine system	Country	Macroinvertebrate studied	Order	Extraction	Observation	Abundance mean (range)	Types	Colors	Polymers	Reference
Danube River	Serbia	*Lithoglyphus naticoides*	Littorinimorpha	KOH, H_2O_2	SM	1.6 ± 0.5 item/organism	Fiber, hard plastic, nylon			Stanković et al. 2022
Danube River	Serbia	*Limnodrilus hoffmeisteri*; *Chironomus acutiventris*	Tubificida; Diptera	KOH, H_2O_2	SM	4.6 ± 1.6; 1.2 ± 0.3 item/organism	Fiber, hard plastic, rubber			Stanković et al. 2022
Braamfontein Spruit Stream	South Africa	*Chironomus* sp.	Diptera	KOH	SM	53.4 pieces/g ww (19.8–96.7)	Filament, angular, round	Blue, black, red		Dahms et al. 2020
Bloukrans River system	South Africa	*Chironomus* spp	Diptera	HNO_3	LM	(0—5.04 p/mg)				Nel et al. 2018
River Höje	Sweden	*Anodonta anatina*	Unionida	HNO_3	LM	75.5 (urban), 41.2 piece/ind	Fiber, particle	Black, green		Berglund et al. 2019
Wu River basin	Taiwan	Chironomidae	Diptera	H_2O_2	SM; SEM	(0.3 ± 0.3–2.1 ± 0.8 piece/mg)	Fiber, film fragment			Lin et al. 2021
River Irwell	UK	*Tubifex tubifex*	Tubificida	KOH	SM; FTIR-ATR	129 ± 65.4 pieces/g	Fiber, fragment	Blue	PES, PP, PVA acrylic/PAN, PE	Hurley et al. 2017
Dorset Stour River	UK	Amphipoda, Annelida, Coleoptera, Ephemeroptera, Diptera, Isopoda, Gastropoda, Hemiptera, Odonata, Megaloptera, Trichoptera		H_2O_2	SM; FTIR		Fragment	Blue/green, Grey/black, Pink/red	PE, PP, PA Polyheptene,	Parker et al. 2022a
Taff, Usk and Wye Rivers	UK	Baetidae, Heptageniidae; Hydropsychidae	Ephemeroptera; Trichoptera	H_2O_2	LM		Fiber			(Windsor et al. 2019)

Table 3 (continued)

Riverine system	Country	Macroinvertebrate studied	Order	Extraction	Observation	Abundance mean (range)	Types	Colors	Polymers	Reference
Bourne Stream	UK	Amphipoda, Annelida, Diptera, Ephemeroptera, Gastropoda, Hemiptera, Isopoda, Odonata, Trichoptera		H_2O_2	SM; FTIR-ATR		Fiber, fragment	Blue/green, pink/red, grey/black	PO, PA, PES, PA	Parker et al. 2022a
Kinnickinnic River	USA	Heptageniidae, Hydropsychidae; Gammaridae	Ephemeroptera; Amphipoda	$FeSO_4+H_2SO_4+H_2O_2$	FM; HNT	(1.2–33.2 particles/g dw)	Fragment, fiber			(Simmerman and Coleman Wasik 2020)

Extraction: *H_2O_2*, hydrogen peroxide; *NaCl*, sodium hydroxide; *KOH*, potassium hydroxide; *HCO_2K*, potassium formate; *HNO_3*, nitric acid. Observation: *DM*, digital microscope; *SM*, stereomicroscope; *LM*, light microscope; *FM*, fluorescence microscope; *HNT*, hot needle test; *μ-FTIR*, micro-Fourier transform infrared microscopy/spectroscopy; *FTIR-ATR*, Fourier transform infrared spectroscopy with attenuated total reflectance; *SEM*, scanning electron microscope. Polymer: *PE*, Polyethylene; *PP*, polypropylene; *SEBS*, styrene ethylene butylene styrene; *ABS*, acrylonitrile butadiene styrene; *CPE*, chlorinated polyethylene; *PES*, polyester; *PA*, polyamide/nylon; *PPA*, polyphthalamide; *PFA*, perfluoroalkoxy alkanes; *PEU*, polyesterurethane; *VE*, vinyl ester resin; *PEC*, polyethylene-chlorinated; *PEI*, polyetherimide; *HDPE*, high-density polyethylene; *PS*, polystyrene; *ECTFE*, ethylene chlorotrifluoroethylene; *PC*, polycarbonate; *PO*, olefin fiber (polyolefin); *PEAA*, polyethylene acrylic acid copolymer; *EVOH*, ethyl vinyl alcohol; *PEA*, polyethylacrylate; *PARA*, polyarylamide; *EPM*, ethylene propylene rubber; *PBA*, polybutylacrylate; *FKM*, fluoroelastomer; *PA 12*, grilamid tr 55; *PTFE*, polytetrafluoroethylene; *BR*, butadiene rubber; *PAN*, polyacrylonitrile; *PVA*, polyvinyl acetate; *P4VP*, poly(4-vinylpyridine); *PDMS*, poly(dimethylsiloxane); *PAA*, poly(acrylic acid); *PAM*, polyacrylamide; *PVC*, polyvinyl chloride; *TPU*, thermoplastic polyurethane; *EPDM*, ethylene propylene diene monomer chemical fragmentation, as plastics are transported downstream (Wicaksono et al. 2021). Most reported MP particles in surface water and sediments are less than 0.5 mm in size (Utami et al. 2021a; Ramaremisa et al. 2022; Saad et al. 2022), with some studies reporting the highest abundance below 0.1 mm (Pan et al. 2021; Parker et al. 2022b). Moreover, such MPs have visible cracks due to friction/abrasion with sediments (such as rocks, gravel, sand, and organic matter) and organisms during transportation through different environments (Wang et al. 2021a). On the contrary, some studies have reported higher abundance of larger MP particles (above size range of 1 mm), especially in surface water (Amrutha and Warrier 2020; Zhang et al. 2020; Nguyen et al. 2022). The presence of large MP materials could be an indication of fresh input of microparticles from the surrounding sources, such as pellets from industries. In the Saigon River, surface water was dominated by pellets with a size range of 2.8–5.0 mm, expected to originate from the surrounding plastic factories (Nguyen et al. 2022).

Regarding MP polymer type in rivers (Table 2), pollution is dominated by PE, PP, PS, PES/PET, PA, and PVC (Lestari et al. 2020; Wu et al. 2020; Xia et al. 2021; Babel et al. 2022; Li et al. 2022). The most common synthetic polymers are PE and PP, used for purposes such as packaging, carrier/shopping bags, agricultural plastic mulch, and fishing nets (Lestari et al. 2020; Xu et al. 2021). Indeed, PS is a common material used in decoration, as well as packaging and transportation, while PVC and PES (PET) are often used in construction, especially plumbing and fabrics/textiles, respectively (Wu et al. 2020; Xia et al. 2021). The high use of these plastic materials increases their input in the environment and eventually aquatic ecosystems. Besides, plastic materials such as PP and PE can easily fragment into microscopic particles, thus increasing their availability in ecosystems (Li et al. 2022). Moreover, high abundance of PES fibers has been reported near textile industries or laundry areas on river banks (Alam et al. 2019). Low-density MP polymers, such as PS, are likely to be suspended on the water surface (Lestari et al. 2020; Wu et al. 2020). However, due to biofouling, these particles can sink to the bottom, and thus be found in sediments.

However, the quantities and characteristics of MP particles reported by different studies are affected by different methodological approaches (Klein et al. 2015; Rodrigues et al. 2018). Although 5 mm is acknowledged and widely reported as the highest size limit of MPs, some studies use varied upper limits. Alam et al. (2019) and Rimondi et al. (2022) used the highest limit of 2 mm, while Said and Heard (2020), Tibbetts et al. (2018), and Kabir et al. (2021) used 4.75, 4, and 1 mm, respectively. Meanwhile, Singh et al. (2021) used a higher "upper size limit" of 7.5 mm. Regarding the "lower size limit," variations are even more, with almost every study considering a different size, perhaps due to limitations in equipment such as nets and microscopes. Analyzing lower-sized MP particles requires more complex equipment, such as high-resolution microscopes and spectrometers which might be inaccessible to many researchers. In many studies, the "lower size limit" of MPs studied is relatively large, above 0.5 mm (Frei et al. 2019; Xu et al. 2021; Nguyen et al. 2022). The abundance of MP reported when a smaller "lower size limit" is used will be high, and its comparison with larger "lower size limit" may not be feasible.

Identifying the polymer types of MPs observed is important as a confirmation of plastic presence and polymer composition. Polymer type is often analyzed using FTIR and Raman spectroscopy and the results used to search for the polymer from spectra libraries (Horton et al. 2017; Battulga et al. 2019; Bian et al. 2022; Khuyen et al. 2022). Other methods used to analyze polymers include Pyr-GCMS (McCormick et al. 2016; Laermanns et al. 2021) and SEM (Ding et al. 2019). Nonetheless, polymer types of the observed MP particles are not analyzed in some riverine studies (e.g., Nel et al. 2018; Ebere et al. 2019; Donoso and Rios-Touma 2020; Kiss et al. 2021). This is partly attributed to the difficulties involved in analyzing of polymers using equipment, such as a micro-FTIR (Talbot et al. 2022). Unfortunately, organic materials, such as cotton, have been reported among observed potential plastic materials during polymer analysis (Alam et al. 2019) and need to be removed from the reported abundance. Therefore, in cases where polymers are not analyzed, MP abundance is likely to be overestimated.

The abundance, distribution, and characteristics of MPs were mainly dependent on the population density and weather/hydrological conditions. Indeed, more MPs have been reported in highly populated and industrialized areas. Generally, MPs increased during the rainy season due plastic influx into the system through surface runoff. Meanwhile, low abundance was recorded in some rivers during the rainy season, which was attributed to dilution of plastic materials. In addition, the method or approach (MP size range) used by different researchers affects the level of abundance reported.

Interaction of macroinvertebrates with MPs

Ingestion of MPs has been reported in many riverine macroinvertebrates (Table 3), such as gastropods (Akindele et al. 2019), Atyids (Nan et al. 2020), bivalves (Wardlaw and Prosser 2020), and aquatic insects (e.g., Dipterans, Ephemeroptera, Odonata, Trichoptera, Coleoptera) (Akindele et al. 2020; Dahms et al. 2020; Lin et al. 2021; Corami et al. 2022). There are two categories of ingestion, i.e., primary and secondary/indirect ingestion. Primary ingestion involves the direct intake of MPs by an organism from the environment, accidentally or intentionally, whereas secondary ingestion occurs when an organism feeds on another with MPs (Lusher et al. 2017; Garcia et al. 2021). Once MPs enter the food web, they can be bioaccumulated and bio-magnified to higher trophic levels (Cuthbert et al. 2019). Feeding strategies and features of individual organisms (such as mouthpart morphology, gut recharge rate, and organism's size), MP abundance, and characteristics (particle density, size, shape, and color) are the main determinants of ingestion and retention of MPs (Scherer et al. 2017; Redondo-Hasselerharm et al. 2018a; Windsor et al. 2019; Gago et al. 2020; Schell et al. 2022). In addition, the exposure pathway influences the ingestion of MPs, i.e., organisms being exposed to MPs in water or sediment (Schell et al. 2022). When exposed to MPs in different pathways, *Hyalella azteca* (Hyalellidae, Amphipoda) and *Asellus aquaticus* (Asellidae, Isopoda) in water ingested more MPs than their counterparts in sediment (Schell et al. 2022).

The ingestion rate of MPs greatly differ among feeding strategies, animal size, and MP size. Scherer et al. (2017) found benthic feeders, such as *Chironomus riparius* (Chironomidae, Diptera), *Gammarus pulex* (Gammaridae, Amphipoda), and *Physella acuta* (Physidae, Gastropoda), to ingest larger particles. In Dommel River (the Netherlands), detritivore Asellidae had the highest MP mean abundance (195 particles/individual), followed by benthic feeder Chronomidae (35 particles/individual), while no particles were reported in predatory Calopterygidae (Pan et al. 2021). Meanwhile, largely detritivorous plecopterans had the highest mean abundance (39.7 particles/individual) in the Rideau and Ottawa Rivers, followed by largely predatory decapods (33.4 particles/individual) (D'Addario 2020). *Chironomus* sp. (Chironomidae, Diptera) are mainly non-selective feeders (Stanković et al. 2020; Silva et al. 2021b), thus intrinsically ingest MP amounts in relation to the environmental concentrations. Indeed, a positive correlation was found between MPs ingested and environmental concentrations; however, they were size specific, with a higher intake of mainly small-sized particles (Silva et al. 2021b).

However, the potential of MP particles to accumulate in the food web increases with decreasing particle sizes (Scherer et al. 2017). Silva et al. (2019, 2021b) found a higher intake of 30 to 60 µm in *C. riparius* (Chironomidae, Diptera), while *Chironomus tepperi* (Chironomidae, Diptera) mainly ingested 10 to 27 µm (Ziajahromi et al. 2018). *Anodontites trapesialis* (Mycetopodidae, Bivalvia) have preference for smaller particles of size range 17 to 88 µm (Moreschi et al. 2020). Meanwhile, *L. costata* (Unionidae, Bivalvia) collected from the Grand River Watershed (Canada) ingested MP particles with a size rage of 21 to 298 µm and an average of 114 µm (SD = 63 µm) (Wardlaw and Prosser 2020) and had MP abundance of up to seven particles/individual. In freshwater mussels, *Anodonta anatine* (Unionidae, Bivalvia), *Sinanodonta woodiana* (Unionidae, Bivalvia), and *Dreissena polymorpha* (Dreissenidae, Bivalvia) MP ingestion are dependent on body size, MP concentrations, and exposure time (Weber et al. 2021a). The body size (weight and length) of aquatic organisms affects their retention of MPs; larger-sized organisms have higher abundance of MPs (Garcia et al. 2021). Further, particle size selectivity increases with body size and growth stage of organisms, i.e., bigger and adult individuals gain the ability to ingest larger particles (Au et al. 2015; Schell et al. 2022). Similarly, larger-sized species tend to ingest larger particles, compared to smaller-sized species. For example, larger mussels *A. anatina* and *S. woodiana* ingested larger MP particles (45 and 90 µm) compared to the smaller *D. polymorpha* (5 µm). As a result, larger organisms are likely to ingest fewer, but larger MP particles.

Relatively high quantities of MPs (mainly small sized) were reported in the gut of *C. riparius* larvae at the fourth (4th) instar (Silva et al. 2019, 2021b). According to Silva et al. (2021b), the high quantities of small-sized MPs in the 4th instar could be due to more MP particles being ingested and a long residence time of MP particles in the guts. Indeed, Scherer et al. (2017) found *C. riparius* larvae to have a low capacity to egest MP particles.

On the other hand, an inverse correlation between MP ingestion and individual body size in *D. polymorpha* was observed; thus, small individuals including juveniles are likely to accumulate more MPs (Weber et al. 2021a). In some bivalves, such as *Mytilus* spp., the consumption rate increases with decreasing body size (Duinker et al. 2007), which can explain the higher MP ingestion in small-sized *D. polymorpha*. Quagga mussels *Dreissena bugensis* (Dreissenidae, Bivalvia) and *A. trapesialis* benthic filter feeders were found to retain up to 95% (Pedersen et al. 2020) and 78% (Moreschi et al. 2020) of the ingested MPs, respectively after 24 h. When exposed to PS MPs, *D. polymorpha* rapidly ingested the materials reaching maximal MP body burden (total number of ingested MPs per individual) within 1 and 12 h (Weber et al. 2021a). Although excretion is relatively fast, MP elimination is not complete, and retained particles are suspected to remain in the gut, translocated to the circulatory system, or body tissue (Weber et al. 2021a). Further, MPs can be retained and transferred from one trophic level to another. In the study by Cuthbert et al. (2019), *Chaoborus flavicans* (Chaoboridae, Diptera) were found to accumulate MP particles from filter feeding *Culex pipiens* (Culicidae, Diptera) its prey, with the rate of transfer significantly related to predation rates.

Denser MP particles, such as PET and PVC, sink to the bottom of aquatic ecosystems and are often ingested by deposit feeders like collector-gatherers. On the other hand, less-dense particles, like PE and PP that float, are often ingested by filter feeders (Ziccardi et al. 2016). However, due to weathering and biofouling, which affect their buoyancy, these particles can sink and become available to deposit feeders. Akindele et al. (2020) found that collector-gatherers, *Siphlonurus* sp. (Siphlonuridae, Ephemeroptera) and *Chironomus* sp., accumulated more MPs than predators, *L. viridis* (Lestidae, Odonata). Due to direct feeding on MP in the sediments, endobenthic *Lumbriculus variegatus* (Lumbriculidae, Lumbriculida) accumulated more MPs than epibenthic *A. aquaticus* and *H. azteca* with less contact to sediment MPs (Schell et al. 2022). Indeed, MPs have been found to get buried and accumulate in sediments, increasing the encounter rate by endobenthic organisms (Leiser et al. 2021; Huang et al. 2022). In contrast, Scherer et al. (2017) found epibenthic *C. riparius* larvae to accumulate more MPs than endobenthic *L. variegatus*, probably implying the presence of other factors, such as exposure time, ingestion and egestion rate, and concentrations in different benthic zones (epibenthic and endobenthic), that interact to influence MP accumulation. Since sediments tend to accumulate more MPs than surface water and water column, benthic organisms are at a higher risk of ingesting and accumulating more MPs than those on the water surface and the column (Schell et al. 2022). Meanwhile, free-swimming organisms, such as bivalve planktonic larvae, can accumulate more MPs than sedentary organisms (Setälä et al. 2016).

Generally, primary consumers often feeding on the particulate matter are more likely to ingest MP particles (Scherer et al. 2017). Meanwhile, MP particles can adsorb on the surface of aquatic plants. For example, PS and PE strongly adsorb on the surface of duckweed *Lemna minor* (Araceae, Alismatales), nonetheless with no effect on plant growth and photosynthesis (Mateos-Cárdenas et al. 2019, 2022). The attachment of MPs to plants increases their bioavailability to primary consumers, such as *G. duebeni*, hence increasing the risk of bioaccumulation and biomagnification. Further, the shape of MP particles is also important in determining their rate of intake. For instance, Wang et al. (2022) reported higher ingestion and egestion of irregular fragments than spheres in the freshwater snails, *Biomphalaria glabrata* (Planorbidae, Hygrophila). Besides, the quantity of ingested MP particles by an organism is mainly dependent on environmental concentrations (Au et al. 2015; Imhof and Laforsch 2016; Scherer et al. 2017; Khosrovyan et al. 2020; Pedersen et al. 2020; Weber et al. 2021b). Other factors affecting ingestion include the exposure time of an organism (Moreschi et al. 2020; Schell et al. 2022), food availability (Weber et al. 2021a), feeding behavior, and body size (Scherer et al. 2017).

Although ingestion and absorption are the most commonly known and widely studied forms of interaction between macroinvertebrates and MPs, there are other forms of interaction with various effects. Trichoptera (caddisflies) incorporate MP particles in their larval cases (Ehlers et al. 2019; Gallitelli et al. 2021), with reported 59% of individual caddisfly cases having an average of 1.14 ± 0.28 MP/larval case (Ehlers et al. 2019).

Effect of MPs on macroinvertebrates

Effects of MPs on feeding, growth, and reproduction

Based on laboratory experiments, the exposure of macroinvertebrates to MPs has been reported to affect organisms through reduced food intake, damage to the digestive tract, and emergence of new individuals. Ingested MPs can move through the digestive system of the organism and be excreted with minimal or no physical and physiological consequences. Khan et al. (2019) found *H. azteca* to have a gut clearance time of 24 to 48 h after exposure to tire tread particles. Likewise, *D. polymorpha* excreted most of the PS MPs within 12 h of ingestion (Weber et al. 2021a). However, not all ingested MP particles are excreted; some remain in the body of the organism, making them vulnerable to physical harm (Klein et al. 2021; Weber et al. 2021a). Their accumulation inhibits nutrient absorption, reproduction, growth, moulting, feeding, survival, and enzymatic changes. Moreover, there are chances of fragmentation of these MP particles within the organism's digestive tract. Indeed, up to 70% fragments were observed in *G. duebeni* after exposure to spherical PE and increased with feeding time (Mateos-Cárdenas et al. 2022). Fragmentation of ingested MPs increases the number of plastic particles within the digestive tract of organisms, increasing the risk of harm and bioaccumulation. Consequently, MPs accumulated in the gut reduce space for ingested nutritive food materials and their food processing (Silva et al. 2019; Stanković et al. 2020).

In the larvae of *Sericostoma pyrenaicum* (Sericostomatidae, Trichoptera), increased MP particles reduced feeding rate (López-Rojo et al. 2020). Likewise, PA microfibers significantly reduced food assimilation in *G. fossarum* after a chronic exposure of 28 days (Blarer and Burkhardt-Holm 2016). When exposed to MPs, bivalves accumulated the particles on their gills and digestive tract (Moreschi et al. 2020; Pedersen et al. 2020). In *D. bugensis*, the filtration rate of food particles reduces with increasing concentration of MP particles on the gills (Cole et al. 2013; Pedersen et al. 2020), impairing their feeding. Moreover, MPs cause tissue damage in the bivalves' gills causing cilia degeneration with increasing concentration (Fu et al. 2022).

Uptake of MPs may also affect the growth and reproduction of macroinvertebrates. For instance, the growth rate of *Gammarus fossarum* (Crustacea, Amphipoda) reduced after the ingestion of polyhydroxybutyrate and polymethyl methacrylate (PMMA) MPs (Straub et al. 2017). Similarly, the growth rate and reproduction of *H. azteca* (Crustacea, Amphipoda) decreased after ingestion of PE MPs after a 42-day chronic exposure (Au et al. 2015). Moreover, the growth rate of *G. pulex* is reduced due to exposure and ingestion of PS MPs, which is attributable to slow egestion, leading to energy reserve depletion (Redondo-Hasselerharm et al. 2018b). A similar effect was found in *H. azteca* when exposed to PP microfibers; however, this was mainly attributed to the longer residence time of microfibers in the guts (Au et al. 2015).

Ingestion of MP particles increased mortality in *Hydropsyche pellucidula* (Hydropsychidae, Trichoptera) by up to 75% (Piccardo et al. 2021) and 50% in *D. bugensis* after a 25-day exposure to HDPE at a high concentration of 0.8 g/L (Pedersen et al. 2020). At a concentration of 500 particles/kg, PE MPs affected the growth, survival, and emergence of *C. tepperi* (Ziajahromi et al. 2018). Exposure and ingestion of PE by *C. riparius* delayed larval growth and emergence of imagoes (Silva et al. 2019). Nevertheless, in both *C. tepperi* and *C. riparius*, the effects were dependent on the size and concentrations of MP particles (Ziajahromi et al. 2018; Silva et al. 2019). A mixture of PVC, PA, PE, PET, PP, and PS MPs negatively affected *C. riparius* larvae, mainly at the early stage (Stanković et al. 2020). On the other hand, ingestion of PS MPs reduces larval emergence and survival of adult *C. riparius* (Varg et al. 2021). In the nematode *Caenorhabditis elegans* (Rhabditidae, Rhabditida), ingestion of PS microbeads (0.1–10.0 μm) inhibited reproduction, i.e., the number of offspring per adult organism (Mueller et al. 2020). Further, the effects of PS microbeads on *C. elegans* negatively correlated with particle size (Mueller et al. 2020). Nonetheless, smaller particles can easily cross the biological membranes and cause intracellular damage (Von Moos et al. 2012; Jeong et al. 2017).

The presence of other stressors in the environment increases the risk of effects of MPs. The toxicity of PE and PS particles in *C. riparius* increases with acute food shortages, as well as a reduction in temperature (Silva et al. 2021b, 2022a; Varg et al. 2021). At 15 °C, exposure of *C. riparius* to PE led to delayed development, as well as a reduction in larval growth and adult weight (Silva et al. 2022a). In *Allonais inaequalis* (Naididae, Tubificida), exposure and ingestion of PE MPs at high temperatures (29 °C) increased reproduction (number of new individuals), but it decreased in the absence of sediments at a lower temperature (19 °C) (Castro et al. 2020). In an experiment by Scherer et al. (2017) to investigate the effect of fine particulate materials (diatomite and kaolin) and PVC MPs on *C. riparius*, chronic exposure to PVC MPs with diatomite kaolin reduced their weight and emergence. Also, the presence of PVC microparticles in the environment enhanced the toxicity of imidacloprid to *C. riparius*. Assessment of the effect of MPs and climate change (increase in temperature and carbon dioxide) on a neotropical shredder *Phylloicus elektoros* (Calamoceratidae, Trichoptera) larvae showed that both factors increased mortality of the larvae, albeit with no evidence of interactive effect (Firmino et al. 2023).

Although detrimental effects from exposure and ingestion of MPs by macroinvertebrates are widely reported, some organisms have shown resilience, mild effects, or no recognizable effects. For instance, the pond snail *Lymnaea stagnalis* (Lymnaeidae, Gastropoda) ingested high amounts of MPs during a 28-day exposure to 100,000 particles/mL of PS fragments, and no significant effect on reproduction, survival, oxidative stress, and energy reserves was registered (Weber et al. 2021b). Similarly, growth in pond snails (*L. stagnalis*) was not affected by the ingestion of PA (Horton et al. 2020). The resilience to MP stress by pond snails can be partially attributed to their adaption to ingesting non-digestible materials naturally occurring in the environment (Weber et al. 2021b). Exposure to PS MPs did not have a lethal effect on the mud snail *Potamopyrgus antipodarum* (Tateidae, Littorinimorpha) (Romero-Blanco et al. 2021). Further, exposure of *P. antipodarum* to a mixture of PVC, PET, PC, PA, and PS MPs did not increase mortality (Imhof and Laforsch 2016). The plastic mix did not affect morphological characteristics (i.e., weight, shell length, and width), the number of developing embryos in the brood chamber, and released juveniles. However, an increased percentage of shell-less individuals was observed, with a higher MP proportion (70% plastics in food), which was attributed to reduced shell development due to low energy assimilation in the presence of elevated MP concentration (Imhof and Laforsch 2016). Meanwhile, the weight or growth of *G. duebeni* (amphipod) was not affected by ingestion of MPs from duckweed *L. minor*; however, mortality increased with longer periods of exposure (Mateos-Cárdenas et al. 2022).

A 28-day exposure of *L. variegatus* to PE (Silva et al. 2021c) and PS (Redondo-Hasselerharm et al. 2018a) did not have a significant effect on its reproduction. Exposure and ingestion of PET did not cause significant mortality in amphipod *H. azteca* (Queiroz et al. 2022). Similarly, acute and chronic exposure of *H. azteca*, *A. aquaticus*, and *L. variegatus* to PES fibers and tire tread particles did not significantly affect their survival (Schell et al. 2022). Chronic exposure (48 days) of *G. pulex* to PET MPs did not have a significant effect on its feeding activity, metabolism, molting, and survival (Weber et al. 2018). Moreover, exposure of *B. glabrata* to PE microbeads (fragments and spheres) from facial scrub did not have a significant effect on their survival and growth (Wang et al. 2022). Further, Redondo-Hasselerharm et al. (2018a) found no adverse effects of particles from car tire treads (combinations of styrene butadiene rubber (SBR) and polyisoprene) on feeding rate, growth, and survival of *A. aquaticus* and *G. pulex* when exposed to particles (10–586 μm) for 28 days. Additionally, the exposure did not affect the growth of *L. variegatus* and *Tubifex* sp. (Redondo-Hasselerharm et al. 2018a; Schell et al. 2022). Likewise, ingestion of PE MPs by aquatic worm *A. inaequalis* had no significant mortality registered at short-term

(96 h) exposure (Castro et al. 2020). Generally, the ingestion of MPs by macroinvertebrates can have detrimental effects on feeding, growth, reproduction, and survival. The effects mainly depend on their type, size, and concentration in the body, as well as the feeding behavior of macroinvertebrates. However, some macroinvertebrates can resist the presence of MPs in the environment mainly due to their adaptation to the non-digestible particles that often exist in their habitats.

Effects of MPs on the physiology/metabolism of macroinvertebrates

Ingestion of MPs by macroinvertebrates diverts energy from growth, reproduction, and emergence to energetically demanding processes, such as immune responses due to mechanical and physiological damage. Indeed, MP ingestion has been reported to cause oxidative stress and trigger enzymatic changes in many freshwater organisms (Guimarães et al. 2021; Silva et al. 2021c; Fu et al. 2022). In *Corbicula fluminea* (Cyrenidae, Bivalvia), ingestion of PS significantly enhanced energy supply-related enzymes, such as catalase, superoxide dismutase (SOD), and glutathione reductase, whereas acetylcholinesterase (AChE) was inhibited (Fu et al. 2022; Hamed et al. 2022). Acetylcholinesterase (AChE) is a neuronal enzyme important in regulating neurotransmission by hydrolyzing acetylcholine, a neurotransmitter (Wiesner et al. 2007; Smina et al. 2016; Kim and Lee 2018). Accumulation of acetylcholine in the synapses due to inhibition of AChE is fatal to the organism (Wiesner et al. 2007; Smina et al. 2016). Ingestion and accumulation of PS MPs increased antioxidants (i.e., SOD) in dragonfly *Aphylla williamsoni* (Gomphidae, Odonata) larvae while AChE activity decreased (Guimarães et al. 2021). In contrast, Parra et al. (2021) reported a significant reduction of SOD activity in the gills and gonads of *C. fluminae* after 7-day exposure to PS, suggesting induced toxicity in the organs. Indeed, catalase activity in the clam's gills increased to counteract the likely oxidative damage from MP toxicity (Parra et al. 2021). Exposure of *L. variegatus* (Annelida) to PE leads to activation of detoxification and antioxidant mechanisms, i.e., increased total glutathione and glutathione-*S*-transferase activity to prevent lipid peroxidation (oxidative damage) and cope with reactive oxidative stress, ROS (Silva et al. 2021c). This was a response to reduced energy reserves, such as sugars and lipids, caused by the ingestion of MP particles. In dragonfly *A. williamsoni* larvae, ingestion of PS nanoplastics caused REDOX imbalance, with an increase in nitric acid and lipid peroxidation (oxidative stress biomarkers), as well as antioxidant activity (Guimarães et al. 2021).

Ingestion of PE MPs in *C. riparius* activated the phenoloxidase (PO) system (Silva et al. 2021a), which is important in the insects' innate immunity for primary response to pathogens (such as bacteria) and parasites (Sideri et al. 2007; Rodriguez-Andres et al. 2012). The activation of the PO system could have been a response to physical damage to the larval gut epithelial cells or oxidative stress caused by MP ingestion (Silva et al. 2021a, b). The injured part of the gut epithelium could be the entry point of pathogens into the main body cavity (Kho and Lal 2018). In *H. azteca* ingestion of PET increased malondialdehyde and significantly reduced glutathione-*S*-transferase (GST) activity (Queiroz et al. 2022). Glutathione-*S*-transferase is a detoxification enzyme, capable of removing harmful xenobiotic and endogenous substances (Chee et al. 2014; Hernandez et al. 2018). High concentrations of PS microparticles caused tissue damage, behavior toxicity, and oxidative stress in *C. fluminea*, inhibiting siphoning and excretion (Parra et al. 2021; Fu et al. 2022).

Large MP particles and accumulation of small-sized particles in *C. riparius* larvae obstructed the gut, leading to oxidative stress, as well as reduced aerobic energy production (Silva et al. 2021b). Inhibition of glutathione-*S*-transferase (GST) and catalase (CAT) occurred in *C. riparius* larvae exposure to larger PE. Acetylcholinesterase (AChE) activity increase in *Culex quinquefasciatus* (Culicidae, Diptera) larvae after ingestion of PE MPs, a likely response to physical injuries and oxidative stress (Malafaia et al. 2020). In addition, a decrease in lipid reserves was attributed to energy costs arising from physical stress and inflammation from ingestion of MP particles, as well as digestion of unprofitable, non-nutritive MP particles (Silva et al. 2021a). Lipid reserves in many invertebrates are minimal; thus, reduction in the intake of nutritive particles can threaten their survival through increased mortality, as well as reduced growth and reproduction (Ayukai 1987). Indeed, PE MPs have been found to have a significant effect on the nutritional status of *C. quinquefasciatus* with an observed reduction in total proteins and carbohydrates, causing changes in metabolism (Malafaia et al. 2020). Accumulation of MPs in the digestive tract can induce an inflammatory response resulting from physical damage to the gut lumen.

Effects of MPs on the behavior of macroinvertebrates

Ingestion and interaction with MPs can promote behavioral changes in macroinvertebrates, such as burrowing, case building (in caddisflies), and locomotion. The mayfly *Ephemera danica* (Ephemeridae, Ephemeroptera) preferred burrowing in substrates contaminated with MPs (ABS, PET, PP, PS, and PVDF) to natural substrates (Gallitelli et al. 2021). However, burrowing in MP-contaminated substrate could potentially increase the rate of ingestion with associated physical harm and physiological changes. In *P. acuta*, ingestion of PS MPs significantly reduced its locomotion (Kumari et al. 2023). Reduction of locomotion can affect the organism's search for food, escape from predators, and pursuit of mates in dioecious individuals. In contrast, locomotion of *C. riparius* larvae increased after ingestion of

PU-MPs, attributable to the lower weight gained from MPs compared to the natural food materials (Silva et al. 2022b).

When exposed to MPs after removing of caddisfly cases, *O. albicorne* (Odontoceridae, Trichoptera) incorporated MPs to rebuild their cases (Gallitelli et al. 2021). Likewise, caddisfly *L. basale* (Lepidostomatidae, Trichoptera) built portable tube-like (normal) cases as well as "emergency cases" (for immediate protection), using PET and PVC particles exposed to them (Ehlers et al. 2020). Although caddisflies can utilize MPs to construct cases, the stability of the cases reduces with increasing MP concentration. The reduced stability is attributed to the difficulty in attaching silk to plastic particles, compared to natural mineral grains (Ehlers et al. 2020). Unfortunately, unstable cases increase the risk of the organism to predation and physical injuries. Since plastics are lighter than sand and other natural materials, cases constructed using MPs make the animals more susceptible to wave erosion (Ehlers et al. 2020). To meet the weight demand of the caddisfly and reduce the risk of erosion, a higher volume of MP materials is needed. Indeed, *O. albicorne* used a significantly higher volume of MP materials to construct a case with similar weight, as those from natural materials (Gallitelli et al. 2021).

Generally, toxicity of MPs in macroinvertebrates is caused by the reduction of dietary intake and interference in food processing due to the ingestion of MP materials (Au et al. 2015; Scherer et al. 2017). Nevertheless, non-digestible particles naturally exist in the environment and are often ingested by macroinvertebrates, especially during food scarcity and starvation. Indeed, adaptation to ingestion of non-digestible materials, like sand, was reported as the main reason for the less effect of MPs observed in detritivorous shredders (Weber et al. 2018). The shredder *G. pulex* covers non-digestible materials with peritrophic membranes to protect the digestive tract from potential damage (Weber et al. 2018). The bioavailability of intrinsic chemicals such as zinc in car tires increases with time as the plastic materials degrade in the environment (Redondo-Hasselerharm et al. 2018a). As the global population is rising, the plastic demand, production, and use are increasing and so is its release into the environment. As such, the exposure of organisms to MP particles is increasing, risking their survival as well as biomagnification in the food web.

Risk of secondary toxicity posed by MP ingestion

Microplastics can act as vectors that harbor and transfer hazardous hydrophobic organic chemicals (HOCs), such as polybrominated diphenyl ethers (PBDEs), polychlorinated biphenyls (PCBs), organochlorine pesticides (OCPs) (e.g., dichlorodiphenyltrichloroethane, DDT), PAHs, and dioxins into the environment, leading to secondary toxicity (Ziccardi et al. 2016; Windsor et al. 2019; Campanale et al. 2020a). These HOCs can be concentrated (sorbed) on MPs and transferred from lower to higher trophic levels (Teuten et al. 2009). Moreover, toxic chemicals are deliberately used as additives during plastic manufacture to enhance the properties of the products (Campanale et al. 2020a). The common toxic substances used as additives include bisphenol A (BPA), brominated flame retardants, phthalates, aluminum oxide, zinc borate, alkylphenols, and antimony oxide (Ziccardi et al. 2016; Campanale et al. 2020a). For instance, flame retardants, plasticizers, dyes, stabilizers, and surfactants are used to reduce flammability, make materials flexible, give color, increase durability, and modify surface properties, respectively (Wagner and Lambert 2018). Plastic additives such as calcium stearate ($CaSt_2$) and zinc stearate ($ZnSt_2$), used as stabilizers to increase plastic transparency and lubricity, have been detected in water and sediments of Maoshou River in Hong Kong (Wu et al. 2020).

Certainly, MPs may release their additives into the water (Piccardo et al. 2021). Exposure of *Chironomus sancticaroli* (Diptera, Chironomidae) to PBDEs induced changes in neuronal enzymes (Palacio-Cortés et al. 2017). Indeed, 2,2′,4,4′,5-pentabromodiphenyl ether (BDE-99) and 2,2′,4,4′-petrabromodiphenyl ether (BDE-47) congeners reduced AChE activity in the fourth instar. Also, PBDEs induced glutathione-*S*-transferase (GST) activity in the larvae. Heavy metals, PAHs, pesticides, and phthalates have been found to increase cell stress response and delay the emergence of male adults in *C. riparius* (Arambourou et al. 2019). Bisphenol A (BPA), an intrinsic MP additive, and phthalates (often incorporated in PVC) are known to cause endocrine disruption in invertebrates even at low concentrations (Cole et al. 2011). Meanwhile, di (2-ethylhexyl) phthalate (DEHP), often incorporated in PVC and PET, causes deformities in the mouthpart of chironomids (Park and Kwak 2008; Keresztes et al. 2013). Hanslik et al. (2020) reported the transfer of a significant amount of dissolved benzo(*k*) fluoranthene (BkF_{aq}) sorbed on PMMA MPs from *C. riparius* to zebra mussels. Ingestion of *C. riparius* exposed to BkF_{aq} increased the activity of 7-ethoxy-*O*-resonifin deethylase (EROD) in the liver tissue of the zebra mussel. The proximity of macroinvertebrates to plastic materials increases the risk of toxic chemicals released from these materials.

Ecological concerns about MP pollution

The presence of MP particles in sediments has been found to cause changes in the macrobenthic invertebrate communities, especially grazers and deposit feeders (Silva et al. 2022a). The growth, survival, and emergence (into adults) of some macroinvertebrates such as *C. tepperi* can be compromised by MP concentrations of above 500 particles/kg (Ziajahromi et al. 2018). In many riverine sediments, MP concentrations of more than 500 particles/kg have been reported. For example, up to 51,968, 63,112, and 149,350 have been reported in rivers in Yangtze in China (Li et al. 2022), Scheldt in Belgium (De Troyer 2015), and Liangfeng in China (Xia et al. 2021), respectively. Thus, the environmental concentration of MPs in some riverine ecosystems poses a threat to sensitive organisms like *C. tepperi* and *S. pyrenaicum* (Ziajahromi et al. 2018; Firmino et al. 2023). Since macroinvertebrates are key in the food web, a threat to one taxon (or group of organisms) will most likely affect the whole food web and the biodiversity in the areas/ecosystem.

The availability of live food resources (e.g., *D. magna*) for macroinvertebrates and other aquatic organisms is often affected by MPs (Aljaibachi and Callaghan 2018). The role of macroinvertebrates in the food web is an important factor that requires thorough investigation, as they can transfer plastic materials to different trophic levels within aquatic and terrestrial communities. With increasing MP pollution in aquatic environments and observed preferences of MP substrate over natural substrates by some macroinvertebrates, the integrity of riverine ecosystems will greatly be compromised. Macroinvertebrates are an integral part of the food web, representing all forms of functional feeding groups in the aquatic food web; hence, efforts should be made to prevent MPs from polluting their habitats.

Macroinvertebrates as potential indicators of MP pollution in riverine ecosystems

The concentrations of MPs in macroinvertebrates and caddisfly cases can be used in the monitoring of MP pollution in aquatic ecosystems. Besides, the amount of MPs ingested depends on their environmental concentrations. The ability of some organisms to retain MPs in a short period, as observed in *D. bugensis* and *A. trapesialis* (Moreschi et al. 2020; Pedersen et al. 2020), makes them good candidates for bioindication of MP pollution in aquatic ecosystems, especially for short-term pollution. In addition, MPs can be detected from their feces and pseudofeces, which can be collected and analyzed without killing the organisms (Moreschi et al. 2020). In addition, some bivalves, such as the invasive clam *C. fluminae*, inhabit a wide range of aquatic environments, making them good representative species for biomonitoring. Further, the high sensitivity of an organism to the MP pollution makes them better candidates for the monitoring of MPs. For instance, *S. pyrenaicum* is highly sensitive, and an increase in MP concentration increases their mortality (Firmino et al. 2023).

Chironomus sp. and *Siphlonurus* sp. larvae (collector-gatherers) have been recommended as potential indicators of MPs in the environment (Nel et al. 2018; Akindele et al. 2020; Stanković et al. 2020). The quantity of MPs ingested by these collector-gatherers correlates with environmental concentrations and is higher compared to other organisms such as predators (Nel et al. 2018; Akindele et al. 2020). Additionally, chironomid larvae can inhabit both the water column and sediments (Kalman et al. 2023), making them a potential indicator of pollution in both aquatic zones. Meanwhile, some macroinvertebrates are good candidates for biomonitoring, since they can be attracted by color of MP particles. For example, caddisflies (*O. albicorne*) and mayflies (*E. danica*) were found to be attracted to MP particles for case building and burrowing, respectively (Gallitelli et al. 2021).

Although macroinvertebrates can be good indicators of MP pollution in the aquatic ecosystems, some taxa, such as blackworms, may not be good candidates due to their behavioral and physiological traits. Blackworm *L. variegatus* ingest MP particles when exposed; however, the accumulated MPs do not correspond to the environmental concentrations. Besides, black worms do not show oxidative damage even after exposure to significantly higher amounts of MPs (Silva et al. 2021c). The use of macroinvertebrates in biomonitoring of MP pollution in riverine and other aquatic ecosystems can be limited by their ability to withstand other organic and inorganic pollutants.

Microplastic pollution in African riverine ecosystems and macroinvertebrates: what is known

Research on MP pollution in African freshwater ecosystems is relatively new, compared to other regions and marine environments, despite the increasing production and use of plastic materials. Indeed, information on MPs in African riverine ecosystems is limited. However, some research findings have been documented, especially in South Africa (Nel et al. 2018; Dahms 2020; Dalu et al. 2021; Ramaremisa et al. 2022). The mean MP abundance in sediments of streams surrounding Lagoon Bizerte in Tunisia ranged from 2340 ± 227.15 in Khima stream to 6920 ± 395.98 particles/kg dw in Jedara stream (Toumi et al. 2019), attributed to pollution from increasing anthropogenic activities in the catchment. The areas surrounding Lagoon Bizerte are densely populated, with streams receiving wastes from municipal, agricultural, and industrial activities including textiles, plastic processing, petrochemicals, ship building, and ship repair (Toumi et al. 2019; El Mahrad et al. 2020). In surface water, up to 1556 particles/L were reported in the Nwangale River in Imo State, Nigeria (Ebere et al. 2019), the second highest abundance reported in riverine surface waters (globally) after Krukut River in Jakarta, Indonesia. The high abundance of MPs in the Nigerian rivers could be attributed to the increasing use of single-use plastic products, like food packs, cups, PE shopping bags, and sachet water (about 60 million sachets are consumed daily), coupled with indiscriminate waste disposal reported in the country (Dumbili and Henderson 2020). Moreover, MPs have been reported in the Densu River, Ghana (Blankson et al. 2022), River Themi, Tanzania (Kundu et al.

2022), and the Iishana system, Kunene, Kavango, and Oranje rivers in Namibia (Faulstich et al. 2022) at lower abundance than those reported in Tunisia, South Africa, and Nigeria.

Ingestion of MPs by riverine macroinvertebrates has been reported in *Chironomus* sp. from the Ogun river of Nigeria (Akindele et al. 2020), Bloukrans River system (Nel et al. 2018), and Braamfontein Spruit stream (Dahms et al. 2020) of South Africa. Other studied organisms are *Siphlonurus* sp., *L. viridis*, *L. varicus* (Gastropoda; Ampullariidae), and *M. tuberculata* (Gastropoda, Thiaridae) from Osun River, Nigeria (Akindele et al. 2019, 2020). The highest average MP abundance was recorded in *Chironomus* sp. at 291.76 ± 26.55 load/g ww (Akindele et al. 2020). Overall, the most encountered MP types were fibers, fragments, and films; an indication that MP contamination in African riverine ecosystems is mainly from secondary sources. The most common polymers recorded were PE, PES, PS, and PP, often used in making carrier/shopping plastic bags, packaging materials, and beverage bottles (Ebere et al. 2019).

Nevertheless, some of the available studies (e.g., Nel et al. 2018; Dahms et al. 2020; Dalu et al. 2021; Blankson et al. 2022) did not analyze the polymer composition. Ebere et al. (2019) analyzed polymer composition using physical characteristics, such as surface texture, rigidity, elasticity, and transparency. Considering only physical characteristics may lead to an overestimation of MPs in the environment, since some natural materials, such as stones and tree branches, can be mistaken as plastics. Evidently, cotton/cellulose are among natural materials which have been misidentified as MPs (Bertoli et al. 2022), only to be ruled out after proper polymer analysis.

Generally, management of solid wastes, including plastics, is still poor in most African countries, with wastes dumped on roads and trenches or open garbage collection points where they are swept by runoff or wind into the terrestrial environment or waterbodies (Dumbili and Henderson 2020). Although some countries such as Uganda and Kenya put bans or restrictions on the production, importation, exportation, and use of plastic materials, implementation is low or lacking. The implementation failure of plastic policies is partly attributed to the business power and the influence of plastic companies and the manufacturing sector (Behuria 2019). However, other countries like Rwanda have successfully implemented policies, restricting the use of PE (and single-use plastic materials), and efficient plastic waste management, with deterrent penalties for non-compliance (Danielsson 2017; Xie and Martin 2022).

Public health concerns about MP pollution

Humans are exposed to MPs through ingestion, inhalation, and skin contact. Many people are exposed to MPs through the direct use of contaminated water for drinking or consumption of contaminated aquatic organisms. Indeed, MPs have been reported in drinking water sources, like household taps, even after going through conventional water treatment (Ferraz et al. 2020; Semmouri et al. 2022). This is an indication that the current water treatment technologies may not be effective in eliminating MPs from drinking water. Moreover, riparian human communities collect drinking water from rivers, often using the same points used for washing clothes and sometimes automobiles, increasing their exposure to MP particles. In addition, MP particles have been reported in aquatic organisms consumed by humans, such as mussels and fish (Khan et al. 2020; Park et al. 2020; Blankson et al. 2022). Further, MPs have been reported in other food items such as sugar, honey, and beer (Liebezeit and Liebezeit 2013, 2014).

Through ingestion and absorption by macroinvertebrates, MPs are introduced into the food web and can bioaccumulate and biomagnify, harming organisms within the food chain (Hamid et al. 2018). The digestive tract faces the highest risk of MP exposure from the ingestion of contaminated food materials. Indeed, laboratory exposure of human intestinal cells HT-29 to PS particles (3 and 10 μm) increased cell mortality and production of reactive oxygen species which can cause intestinal disorders at prolonged exposure (Visalli et al. 2021). In 2022, Leslie et al. (2022) reported the discovery of MP particles in human blood. Since humans have a closed circulatory system, the presence of MPs in blood portends the possibility of MPs crossing from one system to another. A study by Wang et al. (2021b) showed that the uptake of PS MPs in human kidney proximal tubular epithelial cells (HK-2 cells) can cause kidney malfunction, inflammation, endoplasmic reticulum stress, and autophagy of the cells. Human pathogenic bacteria such as *Pseudomonas* sp. which can cause infections of the lungs and blood have been reported to colonize MPs (Park et al. 2020; Blankson et al. 2022; Khan et al. 2022). As such, increased use of plastic material and contamination of the environment with MPs are likely to expose humans to health risks.

Mitigation measures against riverine MP pollution

Measures to reduce pollution and restore the polluted environments are urgently needed due to MPs' potential toxicity and ability to bioaccumulate and biomagnify in the food web. Some organisms such as bivalves, seagrass, and bacteria have potential for bioremediation of MP pollution. Freshwater bivalve *A. trapesialis* was found to eliminate MP particles from the water column through filtration and transfer them to the bottom as part of the pseudofeces (Moreschi et al. 2020). Indeed, bivalves have been found to modify the benthic environments through the assimilation and transfer of nutrients from the water column to the benthic environments. This shows a high potential of bivalves in the bioremediation of MPs in aquatic environments. However, more studies are needed especially on the mechanisms of pseudofeces collection. Some aquatic plants have shown an ability for use in the bioremediation of MPs through trapping and sinking of the particles. For example, MP adherence to aquatic plants has been reported in *Enhalus acodoides*, *Zostera marina*, and *Thalassia testudinum*, making them important MP traps or sinks (Goss et al. 2018; Huang et al. 2020; Jones et al. 2020). Further, the bacterium *Ideonella sakaiensis* produces enzymes (PETase and MHETase) that hydrolyzes PET into terephthalic acid and ethylene glycol which are environmentally harmless (Palm et al. 2019; Yoshida et al. 2021). As such, this bacterium can be adopted for the bioremediation of PET pollutants in the environment. However, research is essential to facilitate the use of these organisms in bioremediation of MPs.

Since secondary MPs, i.e., fragments and fibers, contribute the largest portion of MP pollutants in aquatic ecosystems, management of macroplastic wastes is key in reducing pollution. Many countries have put up regulations on the production, use, and disposal of plastic materials, especially single-use items such as plastic bags, drinking straws, disposable plates, cups, and spoons. While Germany, Tasmania in Australia, and Denmark banned the use of plastic bags, Ireland levies (taxes) on their use (Xanthos and Walker 2017). Unfortunately, even in countries where the use of some plastic materials is banned, implementation has not been very successful (Behuria 2019). For example, China banned the use of plastic bags with a thickness of less than 25 µm, but they are still in use among retailers (Xanthos and Walker 2017).

Primary MPs, such as microbeads and microspheres, which are originally microscopic are harder to manage. MPs are ingredients in common items, such as personal care products (toothpaste, facial scrub, cleansers, detergent), clothes, and paints (Suardy et al. 2020; Faber et al. 2021; Vassilenko et al. 2021; Wang et al. 2022). Unfortunately, common users are unaware of the presence of plastic materials in these products. Use of these materials should to be banned and/or replaced with organic materials, such as plant extracts for personal care products. Nevertheless, the use of MPs in personal care products has been banned in some countries, such as the UK, the Netherlands, Thailand, USA, China, Ireland, South Korea, New Zealand, France, and Canada (Kentin and Kaarto 2018; Habib et al. 2022; Hoang et al. 2022). In general, there is a need for awareness campaigns aimed at educating the population about the hazard of MPs to ecosystems and humans. In addition, the use of alternative materials to plastics should be promoted, such as personal care products with organic ingredients and organic packaging materials, such as paper.

Conclusion

Pollution of riverine ecosystems with MP materials is globally distributed, with rivers crossing through highly industrialized and urbanized areas at higher risk. Population density, economic development, and hydrological conditions are the main factors affecting MP abundance and distribution in riverine water and sediments. Ingestion of MPs is reported in riverine macroinvertebrates including bivalves, gastropods, dipterans, decapods, and odonates, while trichopterans incorporate them in the cases. Through the macroinvertebrates, MPs can bioaccumulate and bio-magnify in the food web, adversely affecting other organisms. The quantity of MP particles ingested or incorporated in caddisfly cases is mainly affected by the environmental abundance. Laboratory experiments on the effects of MP ingestion on macroinvertebrates have shown a reduction in growth, reproduction, and survival. In addition, MP ingestion causes oxidative stress and enzymatic changes in organisms. In caddisflies, MPs reduce the stability of the cases, increasing the risk of predation and injuries from pressure. While macroinvertebrate exposure and ingestion of MPs have been reported to have detrimental effects, some organisms show mild effects or resilience.

Generally, mitigation of MP pollution and restoration of polluted environments require substantial information. Potential biological approaches include seagrasses like *E. acodoides* and *Z. marina*, which can trap and sink MPs, and bacterium *I. sakaiensis* which can hydrolyze PET into environmentally benign chemicals. Meanwhile, some countries have policies banning the manufacture, importation, and use of plastic materials, especially plastic bags and single-use plastics, a good step toward reducing MP pollution. Other countries like China, UK, Canada, and France banned personal care products with MP ingredients. Currently, some areas especially in developing regions like Africa, the Caribbean, and South America do not have enough data to support and guide policy development. Thus, further research is needed to develop realistic and feasible evidence-based policies on the manufacture, importation, and use of plastic products in these regions.

Despite the ecological (part of the food web) and economic importance (source of human food) of macroinvertebrates, research on their interactions (ingestion, case building) with MPs in riverine ecosystems is still limited. In addition, the effects of MP interaction with macroinvertebrates has only been done on a limited number of taxa, mainly chironomids, amphipods, bivalves, and gastropods. Therefore, more research effort is needed to improve the understanding of the interactions of macroinvertebrates with MPs and its effects.

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