Microplastics in urban runoff: Global occurrence and fate

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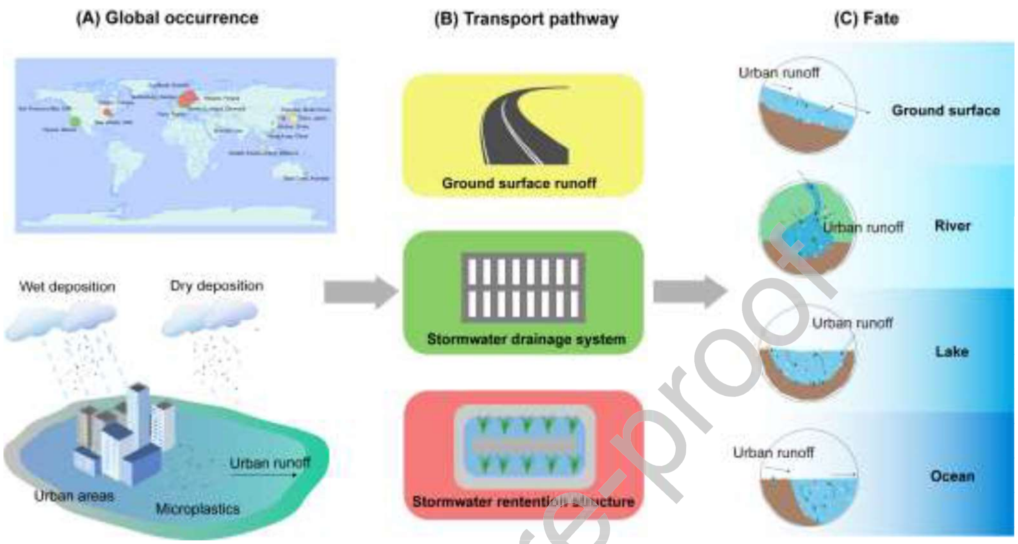
Highlights

- Urban runoff is a major MP transport route from land to aquatic environments.
- MP concentrations in urban runoff were quantitatively assessed.
- Aging, transport, flux and fate of MPs in urban runoff are largely ignored.
- Controlling MP pollution need source prevention and stormwater management.

Graphical Abstract



Abstract

Public concerns on microplastic (MP) pollution and its prevalence in urban runoff have grown exponentially. Huge amounts of MPs are transported from urban environments via surface runoff to different environment compartments, including rivers, lakes, reservoirs, estuaries, and oceans. The global concentrations of MPs in urban runoff range from 0-8580 particles/L. Understanding the sources, abundance, composition and characteristics of MPs in urban runoff on a global scale is a critical challenge because of the existence of multiple sources and spatiotemporal heterogeneity. Additionally, dynamic processes in the mobilization, aging, fragmentation, transport, and retention of MPs in urban runoff have been largely overlooked. Furthermore, the MP flux through urban runoff into rivers, lakes and even oceans is largely unknown, which is very important for better understanding the fate and transport of MPs in urban environments. Here, we

provide a critical review of the global occurrence, transport, retention process, and sinks of MPs in urban runoff. Relevant policies, regulations and measures are put forward. Future global investigations and mitigation efforts will require us to address this issue cautiously, cooperating globally, nationally and regionally, and acting locally.

Keywords: Microplastics; urban runoff; global occurrence; pollution; sink

1 Introduction

Plastic pieces less than 5 mm in size have become a global environmental concern. These pieces, known as microplastics (MPs), have great potential to be found in large quantities across different compartments of the natural environments. Numerous studies have been conducted on their diversified sources, widespread occurrence, environmental persistence, and deleterious effects on the natural environment (Ahmed et al. 2022, Thompson et al. 2004). The potential damage these materials can inflict on marine ecosystems is a particular concern (Bakaraki Turan et al. 2021, Perumal and Muthuramalingam 2022), with research showing that more than 100 marine species, from zooplankton to seabirds, are affected by MP pollution (Cole et al. 2011, GESAMP 2016). In 2011, the 'Declaration of the Global Plastics Associations for Solutions on Marine Litter' was announced, and the signatories included plastics organizations and allied industry associations in 40 countries around the world (Marine Litter Solutions 2019). As the terrestrial environment is the major source (~80%) of marine MP pollution (Hajiouni et al. 2022, Pinon-Colin et al. 2020, Yonkos et al. 2014), curtailing MP migration from land to sea is the highest priority action amongst global efforts to tackle global MP pollution in this Plastic Age (Baho et al. 2021, Rillig and Lehmann 2020).

Key pathways in the transfer of MP pollution to aquatic environments include atmospheric deposition, wastewater effluent discharge, and urban runoff. It is the latter, involving overland flow of water from natural rainfall, snow meltwater, excess stormwater or other sources in urban environments, that has been suggested to be the greatest conduit for MP transfer to the aquatic environment (Fang et al. 2021, Muller et al. 2020, Shruti et al. 2021), as it is attributed to the mobilization and transport of large quantities of land-based MPs and other pollutants (Fahrenfeld et al. 2019, Shruti et al. 2021, Werbowski et al. 2021).

Several recently published articles have further elucidated our understanding of the concentration, characteristics and distribution of MPs in urban runoff, and these findings are among the most interesting. We now know that there are large variances in MP concentrations, complex and diverse sources of MPs, and spatiotemporal heterogeneity of MP pollution in urban runoff (**Table 1**). Strikingly, samples collected in New Jersey, USA, alerted us that MP levels in urban stormwater runoff are much higher than those released in wastewater effluents or blown in the atmosphere (Bailey et al. 2021), and that the annual load of MPs discharged in combined stormwater/sewer systems was six times that of discharge via wastewater treatment plant (WWTP) effluent (Chen et al. 2020) because majority of MPs (up to 97-98%) entered WWTPs are trapped or removed in sludge (Lv et al. 2019, Zhang and Chen 2020). Such findings suggest that urban runoff constitutes a major MP transport route, which demands much greater research attention if we are to stem the migration of MPs.

Other recent studies have estimated local flux levels of MPs transferred from urban to aquatic environments, reporting that urban runoff contributed a significant amount of the MP pollution detected in water courses and bodies. For example, 42% of MPs in European rivers are tire and road wear particles (TRWPs) carried by urban runoff (Siegfried et al. 2017), 43% of MPs in the Warnow estuary, Germany, originate from stormwater systems (Piehl et al. 2021), and 62% of MPs in the Baltic Sea arrived via urban stormwater runoff including sewer overflow (Schernewski et al. 2021).

There remain, however, vast knowledge gaps regarding the occurrence, distribution, aging, fragmentation, transport, flux, fate, and sinks of MPs in urban runoff from a global perspective. Existing reviews within the field have mainly focused on methodology for sample collection, pretreatment, quantification, and identification of MPs in urban runoff, e.g., (Shruti et al. 2021). Sources, environmental behavior and fate of MPs in urban runoff have also been considered, e.g., (Xu et al. 2020). However, a comprehensive critical assessment of the global abundance of MPs in urban runoff and their environmental behavior at this scale has been lacking. Here, we provide a critical literature review that considers: 1) the global occurrence, potential sources, and characteristics of MPs in urban runoff around the world; 2) dynamic changes of MPs in urban runoff during mobilization, aging, transport, and retention processes; 3) the fate and sinks for MPs in urban runoff. The critical review concludes with implications for policy making and future research directions.

2 Global occurrence and potential sources

Urban runoff has been identified as an important land-to-sea pathway for MP migration, supported by the emergence of a number of studies around the world that provide robust quantitative evidence. The studies used in our review were collected from Web of Science Core Collections on February 5th, 2022. We have searched under the advanced research mode with "Topic" as the field tag and the search string was defined as "(TS = (microplastic)) AND TS = (runoff OR urban OR rain OR stormwater OR rainfall OR precipitation OR drain OR flux OR transport OR migration)". A total of 1893 results were preliminarily retrieved from the database ranging from 2000 to 2022, and 957 records remained after excluding some unrelated records. These records were further screened according to the following criteria: (1) These studies should be from peer-reviewed journals; (2) The content is mainly to investigate the concentration, polymer types and other characteristics of microplastics in urban runoff. Finally, a total of 23 studies were identified. We used these published studies to harvest pertinent data from the text, figures, tables or supplementary materials on the abundance, compositions and characteristics of MPs in urban runoff systems, including the ground surface runoff, stormwater drainage systems, and stormwater retention structures. In our review, the ground surface runoff refers to the water flowing on the urban surface due to rainfall, snow, storm, etc., including catchment runoff (Hajiouni et al. 2022) and road runoff (Pinon-Colin et al. 2020). Stormwater drainage system is an important facility for urban runoff discharge (Fuchte et al. 2022) and urban runoff can be sampled at the entrance, inside and outlet of the stormwater drainage systems. Furthermore, stormwater retention structures such as bioretention systems, stormwater retention ponds and constructed wetlands can sustain urban runoff for a period of time and capture some MPs in urban runoff (Lange et al. 2022, Liu et al. 2019, Tan et al. 2022). Therefore, it is very essential to investigate the abundance and characteristics at the entrance, inside, and outlet of stormwater retention structures, thus further understanding the removal performance of stormwater retention structures for MPs in urban runoff (Boni et al. 2022).The extracted data is compiled in **Table 1.** Detailed information of the sampling time, sampling locations, and methods of sampling, pretreatment, quantification and identification of MPs in urban runoff are shown in **Table S1**.

2.1 Occurrence and concentration

Concentrations of MPs in urban runoff can vary greatly (Boni et al. 2022, Shruti et al. 2021). Among the studies compiled in **Table 1**, the concentration varied by three orders of magnitude. The maximum reported concentration was 8580 particles/L (size range from 20-100 μm), collected at the entrance to a stormwater treatment process in Sundsvall, Sweden (Lange et al. 2022). The lowest values were less than 1 particle/L (size range from 25-5000 μm). The distribution of MPs had spatial heterogeneity, which was reflected in the great difference in concentrations in different countries or regions. High levels of MP pollution were found in some regions of Europe, North America and East Asia: 1500-6000 particles/L (size range from 20-5000 μm) of both tire and bitumen microplastics (TBMPs) in Gothenburg, Sweden (Jarlskog et al. 2020), 289 particles/L (size range from 25-5000 μm) in Tijuana, Mexico (Pinon-Colin et al. 2020), 270 particles/L (size range from 10-500 μm) in Viborg, Denmark (Olesen et al. 2019), and 186 particles/L (size range from 106-

5000 µm) in Vaughan, Ontario, Canada (Smyth et al. 2021), 81-292 particles/L (size range from 10-5000 µm) in Tokyo, Japan (Sugiura et al. 2021).

The low abundance of MPs was found in Bushehr, Iran (1.2-3 particles/L, <5000 µm) (Hajiouni et al. 2022), Hong Kong, China (1.4-6.8 particles/L, 37-5000 µm) (Mak et al. 2020), and North of Jutland, Denmark (median 1.41 particles/L, 10-2000 µm) (Liu et al. 2019). The prevalence and concentration differences of MPs in urban runoff might be affected by sampling locations, sampling seasons, sampling and analytical methodology, and human activities (Gonzalez-Ortegon et al. 2022, Li et al. 2018, Zhang et al. 2022b). Currently, our database of MPs in urban runoff is not sufficient for a complete understanding of global concentrations of MPs in urban runoff, as too few researchers have conducted quantitative surveys. For example, the data of MPs in urban runoff in Africa and South America have yet been reported. A global map is urgently needed, especially to estimate fluxes of MPs into freshwater and marine environments, to inform potential management measures. Therefore, we urge more attention to measuring the concentration and distribution of MPs in urban runoff around the world.

Fig. 1 *Global mean concentrations of MPs reported in urban runoff. Information obtained from Table 1. Only the studies of Bailey et al. (2021), Chen et al. (2022), Jarlskog et al. (2020) and Treilles et al. (2021) used the median concentration in Fig.1. The research objects in some areas in Fig.1 included not only MPs, but also other microparticles, as shown in Table 1.*

Table 1. *Microplastics (MPs) in urban runoff*

Sample information			MP concentration (particles/L)			MP characteristics			Reference
Media/system	Location	E × L × P*	Mean	Median	Min-Max	Size (µm)	Shape	Polymer/composition**	
Ground surface									
Catchment	Iran, Bushehr	3×8×1	1.9	--	1.2-3.0	<5000	Fibers, fragments, filament, granules, foam, others	--	(Hajiouni et al. 2022)
Catchment	France, Paris	4×1×3-5	--	29	3.0-129	25-5000	--	PE, PP, PS, others	(Treilles et al. 2021)
Catchment	France, Paris	--	35	40	24-60	--	Fibers	--	(Dris et al. 2018)
Catchment	Canada, Ontario	1×3×3	15	--	2.3-29	125-5000	Fibers, films, fragments, foams, bundles, others	PP, PE, PVC, PTFE, Nylon, PA, PU, PET others	(Grbic et al. 2020)
Catchment	USA, California, San Francisco	1×12×1	8.3	--	1.1-25	125-5000	Fibers, fragments, others	PE, PET, PP, cellulose acetate, copolymers	(Werbowski et al. 2021)
Catchment	USA, California, San Francisco	1×12×1	8.1	--	1.1-24	125-5000	Fragments, fibers, films	PE, PET, cellulose acetate	(Zhu et al. 2021)
Road runoff	Malaysia, Greater Kuala Lumpur	4×1×3	--	27	--	--	Fragments, fibers	--	(Chen et al. 2022)
Road runoff	Mexico, Tijuana	7×3×1-3	167	--	13-366	25-5000	Fibers, fragments, films, granules	PE, PS, PA, PET, PP	(Pinon-Colin et al. 2020)
Road runoff	Tokyo, Japan	1×1×2	186.5	-	81-292	10-5000	Fibers, flakes	AS, EVA, PEPD, PEP, PET, PS, PE, PP	(Sugiura et al. 2021)
Stormwater Drainage									
Entrance	China, Wuhan	3×8×1	9.5	--	2.0-22	37-5000	Fragments, fibers, granules, films	PE, PP, PET, PVC, PS	(Sang et al. 2021)
Entrance	S. Korea, Cheonan	--	8.3	--	--	--	Microdebris	PP, PE, PET, PU, others	(Yano et al. 2021)
Inside	Sweden, Gothenburg	5×1×1	--	4400	1500-6000	20-5000	--	TBMPs	(Jarlskog et al. 2020)
Inside	Sweden, Gothenburg	6×1×1	581	--	98-1485	20-5000	--	--	(Jarlskog et al. 2021)
Inside	Mexico, Tijuana	7×1×1-3	88	--	18-139	25-5000	Fibers, fragments, films, granules	PE, PS, PA, PET, PP	(Pinon-Colin et al. 2020)
Outlet	China, Hong Kong	4×2×3	4.6	--	1.4-6.8	54-	Fibers, fragments, pellets,	PE, PP, nylon 6/6, others	(Mak et al. 2020)

						1000	others		
Outlet	Mexico, Tijuana	7×1×1-3	289	--	12-2054	25-5000	Fibers, fragments, films, granules	PE, PS, PA, PET, PP	(Pinon-Colin et al. 2020)
Outlet	USA, New Jersey	3×2×1	0.3	--	--	250-2000	Fragments, films, granules	COPOLY, PET, PE, PS, PP	
Outlet	USA, New Jersey	1×3×1	--	0.6	0.4-0.6	500-2000	Fragments, pellets, sheets	PE, PS, others	
Stormwater Retention Structures									
Entrance	S. Korea, Cheonan	3×1×1	24	--	--	--	Microdebris	PP, PE, PET, PU, others	(Yano et al. 2021)
Entrance	Finland, Helsinki	3×1×1	29	--	8.0-66	90-5000	--	PE and PP	(Pankkonen 2020)
Entrance	Sweden, Sundsvall	9×1×1	1212	230	42-8580	20-100	--	PP, EVA, EPDM, SBR others	(Lange et al. 2022)
Entrance	Sweden, Sundsvall	9×1×1	196	13	0.4-1624	100-300	Fibers, fragments, granules, others	-	(Lange et al. 2021)
Entrance	Sweden, Sundsvall	9×1×1	3.0	0.2	0.3-23	300-5000	Fibers, fragments, granules, others	-	(Lange et al. 2021)
Entrance	Canada, Ontario, Vaughan	17×1×1	186	--	--	106-5000	Fibers, rubber, films, fragments	PET, PU, PE	(Smyth et al. 2021)
Entrance	USA, California, San Francisco	3×1×1	1.9	--	--	125-5000	Fibers, fragments, others	PET, rubber, PE, acrylic, PU	(Werbowski et al. 2021)
Entrance	USA, California, San Francisco	3×1×1	1.6	--	0.4-3.2	125-5000	Fibers, fragments, others	Anthropogenic microparticles	(Gilbreath et al. 2019)
Entrance	Australia, Queensland	2×1×2	0.9	--	--	25-5000	Fragments, fibers	Poly(styrene-co-ethylacrylate), PP, nylon, PET, PE	(Ziajahromi et al. 2020)
Inside	Denmark, Jutland	1×7×1	6.0	1.4	0.5-23	10-2000	--	PVC, PS, PP, PE, PET, others	(Liu et al. 2019)
Inside	Denmark, Viborg	5×1×1	270	--	--	10-500	Fragments, fibers, others	PP, PET, PS, PA, PE, others	(Olesen et al. 2019)
Outlet	S. Korea, Cheonan	3×1×1	4.2	--	--	--	Microdebris	PP, PE, PET, PU, others	(Yano et al. 2021)
Outlet	Finland, Helsinki	3×2×1	1.9	--	--	90-5000	--	--	(Pankkonen 2020)
Outlet	Sweden, Sundsvall	9×2×1	82	--	0-240	20-100	--	PP, EVA, EPDM, others	(Lange et al. 2022)
Outlet	Sweden, Sundsvall	9×2×1	6.2	--	0-39	100-300	Fibers, fragments, granules, others	--	(Lange et al. 2021)
Outlet	Sweden, Sundsvall	9×2×1	0.3	--	0-1.7	300-	Fibers, fragments,	--	(Lange et al.

						5000	granules, others		2021)
Outlet	Canada, Ontario, Vaughan	14×1×1	31	--	--	106-5000	Fibers, rubber, films, fragments	PET, PU, PE, acrylic	(Smyth et al. 2021)
Outlet	USA, California, San Francisco	3×1×1	0.1	--	--	125-5000	Fibers, fragments	PET, PE, acrylic, polyacrylamide	(Werbowski et al. 2021)
Outlet	USA, New Jersey	3×1×1	0.8	--	--	250-2000	Fragments, films, granules	COPOLY, ABS, PE, PP, PS	(Boni et al. 2022)
Outlet	Australia, Queensland	2×1×2	4.0	--	--	25-5000	Fragments , fibers	Poly(styrene-co-ethylacrylate), PP, nylon	(Ziajahromi et al. 2020)

NOTE: * E × L × P = sample number, as Events x Locations x Parallel samples **Abbreviations: ABS, Acrylonitrile styrene-butadiene; AS, Acrylonitrile styrene; COPOLY, Copolymer of ethylene-ethyl acrylate; EPDM, Ethylene propylene diene rubber; SBR, Styrene butadiene rubber; EVA, Ethylene-vinyl acetate; PA, Polyamide; TBMPs, Tire and Bitumen Microplastics; PE, Polyethylene; PEPD, Polyethylene polypropylene diene; PEP, Polyethylene-polypropylene copolymer; PET, Polyethylene terephthalate; PP, Polypropylene; PS, Polystyrene; PU, Polyurethane; PVC, Polyvinyl chloride; PTFE, Polytetrafluoroethylene.

2.2 Spatiotemporal variability

Temporal changes in MP levels in urban runoff are highly correlated to precipitation events. Rainfall is a key environmental factor in the formation of urban runoff, which washes away loose MPs deposited on the ground surface (Hitchcock 2020, Xia et al. 2020). Increasing rainfall intensity also promotes the mobilization of lodged MPs accumulated on land, thus increasing the amount of MPs transported by urban runoff (Sang et al. 2021). On the other hand, higher intensity rainfall events reduce MP concentrations in urban runoff due to dilution effects. For example, a recent study documented that the concentration of MPs in the stormwater outfall at Kwun Tong Ferry Pier, Hong Kong, was higher at the end of dry season than at the beginning (Mak et al. 2020). Another recent study reported a significant negative correlation between MP concentrations in urban stormwater runoff and cumulative rainfall in the range 1.5-4.5 mm (Boni et al. 2022). Similarly, Sugiura et al. (2021) found that higher abundance of MPs in runoff at first flush stage than transitional and steady state stage during the continuous rainfall event.

Different land use in urban catchments is linked to different sources of MP pollution, spatial variability of MPs and an important factor in the emergence of MPs in urban runoff (Chen et al. 2020, Fang et al. 2021). Several studies have established relationships between MP concentrations in urban stormwater runoff and land use (**Fig. 2a**). Typically, runoff samples collected from highly disturbed residential, commercial, and industrial areas tend to contain more MPs than those collected from less-disturbed areas such as university campuses or open land. Large variation in MP concentration among different residential and commercial areas has been observed (**Fig. 2a**). Concentrations of MPs in urban stormwater drainage in different locations are also strikingly different (**Fig. 1 and Table 1**). These data indicate high heterogeneity and, thus, uncertainty in estimating MP levels in urban runoff.

Morphotype also influences mobilization patterns with time. Treilles et al. (2021) collected stormwater runoff samples in Paris, France, at four different sampling times (**Fig. 2b**). The observed changes in fiber MP concentrations were quite different from those in particulate MPs (e.g., spherical MPs), with a reduction in rainfall intensity from March to May leading to a sharp increase in fiber MP concentration and a decrease in particle MP concentration, suggesting a different mobilization mechanism (**Fig. 2b**).

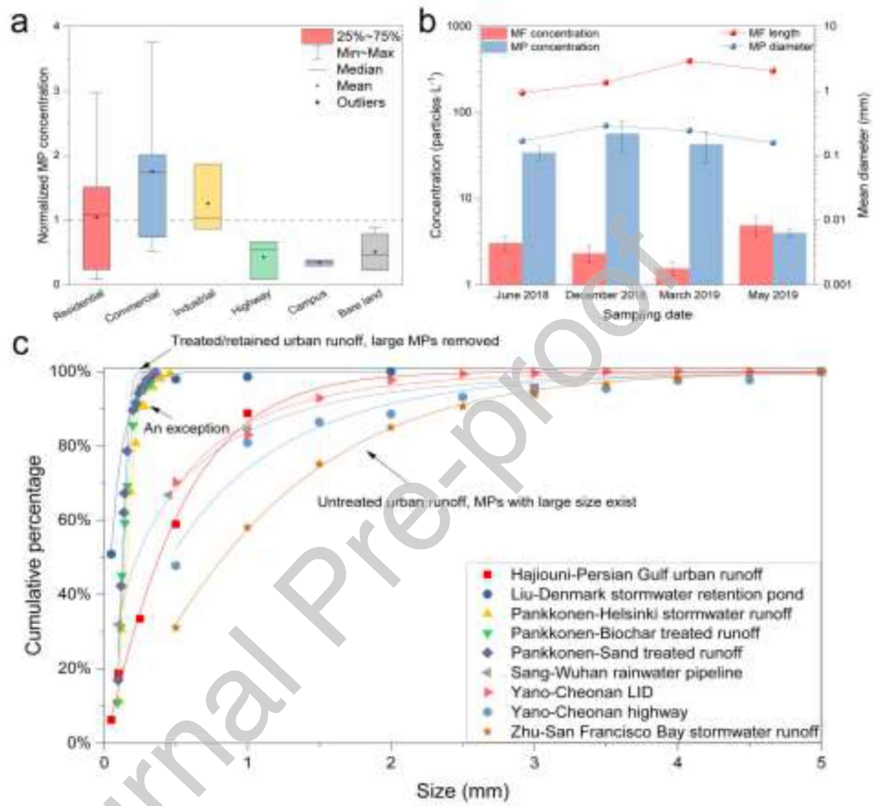


Fig. 2 *(a) Effect of land use type on MP accumulation in urban runoff. Considering that different sampling and analysis strategies applied in different studies lead to high variations of MP concentrations, a normalization of MP concentration according to the average value of a certain study was applied for inter-group comparison. Data source: residential (Liu et al. 2019, Pinon-Colin et al. 2020, Sang et al. 2021, Werbowski et al. 2021), commercial (Liu et al. 2019, Pinon-Colin et al. 2020, Sang et al. 2021), industrial (Liu et al. 2019, Pinon-Colin et al. 2020), highway (Liu et al. 2019, Sang et al. 2021), campus (Sang et al. 2021), bare land (Werbowski et al. 2021). (b) Temporal variability of microfibers (MFs) and particulate microplastics (MPs) in stormwater runoff collected in Paris, France. Data source: (Treilles et al. 2021). (c) Cumulative size distribution of MPs from urban runoff. All data was fitted with a conditional fragmentation model proposed by Wang et al. (2021c) that depicts fragmentation-induced size distribution ($p < 0.05$ for all fittings). Untreated urban runoff samples usually possessed MPs with large sizes, whereas treated or retained ones were enriched with smaller ones. Data source: (Hajiouni et al. 2022, Liu et al. 2019, Pankkonen 2020, Sang et al. 2021, Yano et al. 2021, Zhu et al. 2021).* ***Abbreviations:*** *MFs, microfibers; MPs, microplastics; LID, low impact development.*

2.3 Sources

Unknown and diversified sources of MPs render the concentrations, compositions and characteristics of MPs in urban runoff complex and changeable (Hitchcock 2020, Sang et al. 2021, Shruti et al. 2021) which makes the implementation of management measures more difficult (Fahrenfeld et al. 2019, Grbic et al. 2020). Therefore, it is important to trace the source of MPs in urban runoff.

Of all plastic ever created (Geyer et al. (2017) calculated this number to be ~8.3 billion metric tons), more than 70% has reached its end-of-life, but little has been recycled (~9%) (Geyer et al. 2017). In urban areas, a large amount of plastic waste is poorly managed and released as litter (Fan et al. 2022), with high potential to degrade and become a source of MPs in urban runoff (Clayer et al. 2021, Shruti et al. 2021). In particular, low-income countries rely primarily on open dumping (93%) to dispose of their plastic waste (Kaza et al. 2018), representing a potentially huge source of MPs to the urban runoff.

Road dust is another potentially huge source of MPs in urban areas (Horton and Dixon 2017, Yukioka et al. 2020) and a major reservoir (Aghilinasrollahabadi et al. 2021, Campanale et al. 2022, Monira et al. 2021b) (**Fig. 3**). MPs associated with road dust are mainly derived from tire wear particles (Karbalaei et al. 2018, Monira et al. 2021b), which are formed due to friction effects at the interface between vehicle tires and road surface (Campanale et al. 2022). Abrasion of road markings may also contribute significant MPs to the urban environment (NIPHE 2016). Tire and road wear particles typically consist of ~25% synthetic polymers by mass (Kreider et al. 2010, Unice et al. 2019), and are considered MPs because their physicochemical properties and size range meet the definition of MPs (Campanale et al. 2022). In the EU, tire and road wear particles are reported to form at a rate of ~1 kg per person per year (Kole et al. 2017).

The impact of tire and road wear particles (TRWPs) remains an underexplored area of research. One study stated that they could account for 5-10% of all plastic entering the ocean globally (Kole et al. 2017). A study of road stormwater runoff in Gothenburg, Sweden reported that 220 particles/L (≥ 20 μm) originated from tire wear, while MPs designated to plastic waste (69 particles/L) and road marking paints (14 particles/L) were much lower (Jarlskog et al. 2021). Another study reported a concentration of tire and road wear particles of 5900 particles/L (≥ 20 μm) in road stormwater runoff, suggesting an important source of MPs in urban stormwater runoff (Jarlskog et al. 2020).

Other sources of MPs in urban runoff include industrial activities (Grbic et al. 2020), construction (Shruti et al. 2021), landfill leachate (Golwala et al. 2021, Sulistyowati et al. 2022), laundry of synthetic textile fabrics (Qian et al. 2021), fragments of polymer based paints coatings, as well as MPs suspended in the atmosphere that transfer to surface runoff during rainfall events (Chen et al. 2022, Dong et al. 2021) (**Fig. 3**). Furthermore, the use of plastic rainwater facilities can increase the release of MPs through aging and hydraulic scouring during the collection, transportation and discharge of urban stormwater runoff (Zhang et al. 2022a).

***Fig. 3** Sources of MPs in urban runoff.*

3 Characteristics and dynamic processes

Several studies have investigated the characteristics of MPs in urban runoff (**Table 1 and Table S1**). This section provides a detailed analysis of the characteristics and dynamic processes of MPs in urban runoff.

3.1 Characteristics

Various MP polymer types, i.e., polyethylene (PE), polypropylene (PP), polystyrene (PS), polyethylene terephthalate (PET) and polyvinyl chloride (PVC), have been identified in urban runoff (**Table 1**). These polymers contribute global plastic products in 2020 as PE, 30.3%; PP, 19.7%; PVC, 9.6%; PET, 8.4%; and polyurethane (PU), 7.8%; PS, 6.1% (PlasticsEurope 2021). The large amount of MPs formed from PP and PE polymers is consistent with these being the most produced plastic polymers worldwide (PlasticsEurope 2021). Sang et al. (2021) reported that PE (41.7%) and PP (31.3%) were the main polymer types in rainwater drainage in Wuhan, China. In Paris, France, Treilles et al. (2021) found most (>85%) MPs were composed of PE, PP and PS. In Finland, the total proportion of PE and PP MPs was >99% in urban stormwater runoff (Pankkonen 2020).However, these studies mentioned above did not investigated TRWPs due to the limitations of MP analytical techniques e.g., Raman spectroscopy (Sang et al. 2021, Smyth et al. 2021), and Fourier Transform Infrared spectroscopy (FTIR) (Pankkonen 2020, Treilles et al. 2021). In order to both investigate the commercial MPs (e.g., PE, PP, PVC, etc.) and TRWPs in urban runoff, some studies have adopted micro-FTIR as the identification method of commercial MPs and Attenuated Total Reflection (ATR)-FTIR as the identification method of rubber particles (Lange et al. 2022, Sugiura et al. 2021). Lange et al. (2022) found that black Ethylene propylene diene rubber (EPDM) and PP particles were the most abundant polymer types of 20-100 µm sized MPs in the highway runoff. Sugiura et al. (2021) documented that Polyethylene-polypropylene copolymer (PEP, 34.9%) might derived from ethylene propylene rubber was very common in urban road runoff at first flush stage by rainfall, second only to PE (48.6%).

Furthermore, several technologies e.g., Pyrolysis-gas chromatography-mass spectrometry (Py-GC/MS) and Thermal extraction desorption-gas chromatography-mass spectrometry (TED-GC/MS), are used to specifically detect TRWPs in environment samples (e.g., road dust, sediments, road runoff) (Eisentraut et al. 2018, Goßmann et al. 2021). However, only a few studies have reported the characteristics of TRWPs in urban runoff by using these techniques. A novel study reported that Styrene butadiene rubber (SBR) was found in ranges of 3.9-9.3mg/g in a street runoff grab sample and a sediment sample collected in Halensee, Berlin by using TED-GC/MS, which was much higher than PE, PP and PS. Another recent study documented that the TRWPs in untreated road tunnel wash water (55.3±15.2 mg/L) quantifying by Py-GC/MS was comparable to those in road runoff (Rødland et al. 2022). Although TRWPs are very abundant in urban runoff, their characteristics in urban runoff are largely unknown. The four most common MP morphologies in urban runoff are fragments, fibers, films and granules (Monira et al. 2021b, Paul-Pont et al. 2018) (**Table 1**). For instance, the dominant morphology identified at the inlet of a bioretention rain garden in the San Francisco Bay Area was identified as fibers (58%) by Gilbreath et al. (2019). Similarly, most MPs were considered to be fibers in urban runoff in Ontario, Canada (Grbic et al. 2020), in Vaughan, Canada (Smyth et al. 2021), and in Bushehr, Iran (Hajiouni et al. 2022). The dominance of fibers might be linked to the use of synthetic textile materials and building materials (Carr 2017, Monira et al. 2021b). Different MP colors have also observed, which may indicate the potential sources of MPs (Monira et al. 2021b). For example, a recent study reported that the most common color in urban runoff in Bushehr, Iran, was black (40.2%), which was related to a local fishery plastic (Hajiouni et al. 2022, Martin et al. 2017). Another study identified black MPs specifically as tire and road wear particles (Grbic et al. 2020). However, the polymer types and exact sources of MPs need to be further identified by some analytical techniques (e.g., FTIR and Raman spectroscopy) (**Table S1**).

Distinct MP sizes in urban runoff have been observed among different studies (**Fig. 2c**). Generally speaking, the size range of untreated urban runoff tends to be larger than that of treated or retained ones (**Fig. 2c**), due to filtration (Pankkonen 2020) and settlement (Semcesen and Wells 2021). Several studies that evaluated the effectiveness of stormwater management facilities report a similar phenomenon (Section 4.5). Furthermore, polymer density can affect the distribution of MPs. A recent study reported that the fraction of PET and PVC in pipe sediment was larger than in water (Sang et al. 2021). This was attributed to a high rate of deposition due to the relatively high density of PET and PVC.

The retention of MPs during their transportation process across different environmental compartments via urban runoff (**Fig. 4**) is affected by MP characteristics (e.g., density, size and shape) as well as climatic, topographic and hydrological conditions and human activities (Horton et al. 2017b, Liu et al. 2019, Nizzetto et al. 2016). Further examination of retention mechanisms and the influence MP characteristics is needed to more accurately model MPs emissions through urban runoff.

3.2 Aging and fragmentation

After being released into the environment, plastic debris undergoes a series of physicochemical and biological processes, such as mechanical abrasion, photodegradation, hydrolysis, thermal oxidation, chemical oxidation and biodegradation, resulting in further aging and fragmentation of secondary MPs (Aghilinasrollahabadi et al. 2021, Hanun et al. 2021). During the migration and mobilization of MPs in urban runoff, the interactions between runoff water and MPs e.g., mechanical breakdown (Duan et al. 2021), hydrolysis (Sarno et al. 2020), UV exposure (Alimi et al. 2022) and biodegradation (Hanun et al. 2021), can accelerate the aging process of MPs in urban runoff. Mechanical breakdown is caused by abrasion and disintegration force, which may be the result of the action of water shear force on MPs and the interaction of MPs with sediments, stones, and pebbles in urban runoff (Duan et al. 2021). A recent study simulated the mechanical interactions between synthetic stormwater and MPs, and found that aging processes were related to the transport of MPs from urban environment to water resources (Aghilinasrollahabadi et al. 2021). Another study documented that water shear stress could promote the formation of nanoplastics by crack propagation and crushing mechanism (Enfrin et al. 2020).Hydrolysis is more plausible in some polymers with water sensitive groups in the polymer backbone, such as polyesters (including PET), polyamides, polycarbonates, polyanhydrides, and polyethers (Sarno et al. 2020). Therefore, MPs with sensitive groups are easily hydrolyzed in urban runoff, which further accelerates the aging process of MPs. Furthermore, the retention time of MPs in urban runoff may affect the degree of hydrolysis of MPs. A recent study reported that the hydrolysis of Poly (butylene adipate-co-terephthalate) (PBAT) first occurred on the ester group between the terephthalate and adipate groups, resulting in a decrease in the molar mass (Deshoulles et al. 2022). The subsequent hydrolysis occurred on the ester group within the adipate ester group under longer aging durations (Deshoulles et al. 2022).

In drainage system, fragmentation of plastics and MPs in runoff water likely occurs but lacks quantitative studies. In addition, urban runoff mixes a variety of pollutants from different sources (Muller et al. 2020) making it one of the main sources of water quality damage to aquatic ecosystems (Zhao et al. 2010). Changes in MPs characteristics (e.g., functional groups and hydrophobicity) induced by aging processes can affect their tendency to adsorb other pollutants (e.g., heavy metals and organic contaminants) in urban runoff (Aghilinasrollahabadi et al. 2021, Hanun et al. 2021, Zhang et al. 2022a), further exacerbating ecological risks (Jemec Kokalj et al. 2019, Meides et al. 2021).

Degradation and biodegradation of MPs in environments prior to entry of runoff may occur but no research has been reported on the fate of MPs before and after entry of runoff. In environment, MPs can be generated from plastic waste and/or further degraded via. various abiotic (e.g. photodegradation by UV, thermal degradation by sunlight, chemical catalyzed degradation as well as mechanical degradation) and biotic degradation (Zhang et al. 2021). Photodegradation induced by UV radiation or photooxidation is the most important abiotic aging pathway of MPs in the environment (Alimi et al. 2022). Wang et al. (2021f) found that functional groups (i.e., -OH, C=O and =CH) appeared on the surface of PE MPs during aging by UV light, and found that the carbonyl index increased from 0.07 to 0.62. Another study involving a 13-week rooftop weathering experiment reported the occurrence of substantial aging, suggesting that solar radiation and rainwater were urban environmental stressors for MPs (Miranda et al. 2021).

Biodegradation plays a significant role in the decomposition of MPs in aquatic and terrestrial environments (Luo et al. 2022, Ren et al. 2021). To date, plastics-degrading microorganisms (bacteria, fungi) have been found in soils, sludge, plastic dumping site, sewage, landfill etc. (Ru et al. 2020). In particular, MPs in urban runoff can carry microorganisms (including plastics-degrading microorganisms and pathogens), constituting the microecosystem called ‘plastisphere’ (Junaid et al. 2022, Wang et al. 2022). However, the degradation potential of plastisphere to MPs and ecological risk of disease transmission of plastisphere in urban runoff are largely unknown. In addition, soil invertebrates like earth worms, snails etc. also ingest, defragment and transport plastics and MPs (Huerta Lwanga et al. 2018, Rillig et al. 2017, Song et al. 2020). Especially, the land snails also biodegrade PS via. gut microbial activities (Song et al. 2020). The larvae of several insects e.g., *Tenebrio moliutor* or mealworms which belong to darkling beetles, are capable of biodegradation of PS, PE, PP, PVC etc. via gut microbial activities (Wu and Criddle 2021, Yang et al. 2015). The impact of microbial and invertebrate-based biodegradation on MPs in runoff remains further investigation.

3.3 Transport

Urban runoff is a major vector for the movement of MPs in urban environments (Auta et al. 2017, Fahrenfeld et al. 2019), which is often discharged into nearby water bodies directly or through stormwater drainage and stormwater retention structures (Shruti et al. 2021) (**Fig. 4**).

Rainfall intensity is an important factor in the formation of urban runoff and a driving force for the migration of MPs in urban environments (Fan et al. 2022, Xia et al. 2020), and a recent study inferred that the mobilization of MP occurs when rainfall intensity >2.5 mm/h for more than 2 h (Treilles et al. 2021). In general, the greatest delivery of MPs to rivers occurs during the “first flush” at the beginning of a rainfall event, and as the storm continues, the input will decrease (Barrows et al. 2018, Lee et al. 2002, Wang et al. 2017). It was reported that the abundance of MPs in an urban river in Hong Kong decreased sharply from 14.015 to 1.298 pieces/m^3 within 2 hours due to rainfall-induced runoff (Cheung et al. 2018).

The size of MPs affects their mobility in urban runoff, with smaller sizes (<50 μm) preferentially transported to receiving waters through urban road runoff (Klockner et al. 2020). Another study reported that high-density MPs (e.g., PVC, PET) tend to be deposited in sediment rather than transported (Sang et al. 2021). Nevertheless, the mechanisms and influencing factors in the migration of MPs in urban runoff are largely unknown.

Estimating the contribution of MP migration from land to water bodies via urban runoff is crucial. Siegfried et al. (2017) calculated that tire and road wear particles transported by urban runoff were the largest source of MPs in European rivers (accounting for 42%). Another study indicated that the discharge of MPs from separated city stormwater system accounted for 43% of MPs in the Warnow estuary, Germany (Piehl et al. 2021). Several studies have attempted to estimate the flux of MPs in urban runoff at different temporal and spatial scales through the combination of sample data and mathematical modelling (Bondelind et al. 2020, Clayer et al. 2021).

Bondelind et al. (2020) simulated the distribution of traffic-associated MPs in the Göta River using a three-dimensional hydrodynamic model, finding that MP concentrations were correlated to the number of nearby number of rainwater discharge points and the local traffic load. A recent study estimated that the amount of MPs released from road runoff in the Mjøsa catchment, Norway, is ~5.5 tonnes/year by socio-economic modelling of MP fluxes on a catchment-scale (Clayer et al. 2021). In Shanghai, China, Chen et al. (2020) estimated that the annual load of MPs discharged via drainage systems at the catchment scale is up to 850 trillion particles/year. However, it should be noted that owing to the wide range of sources of MPs in urban areas, the spatiotemporal variation of MPs in urban runoff and sampling errors, it remains difficult to fully quantify flux levels (Bai et al. 2022, Clayer et al. 2021). Therefore, the urban runoff component of the global MP budget is not well understood.

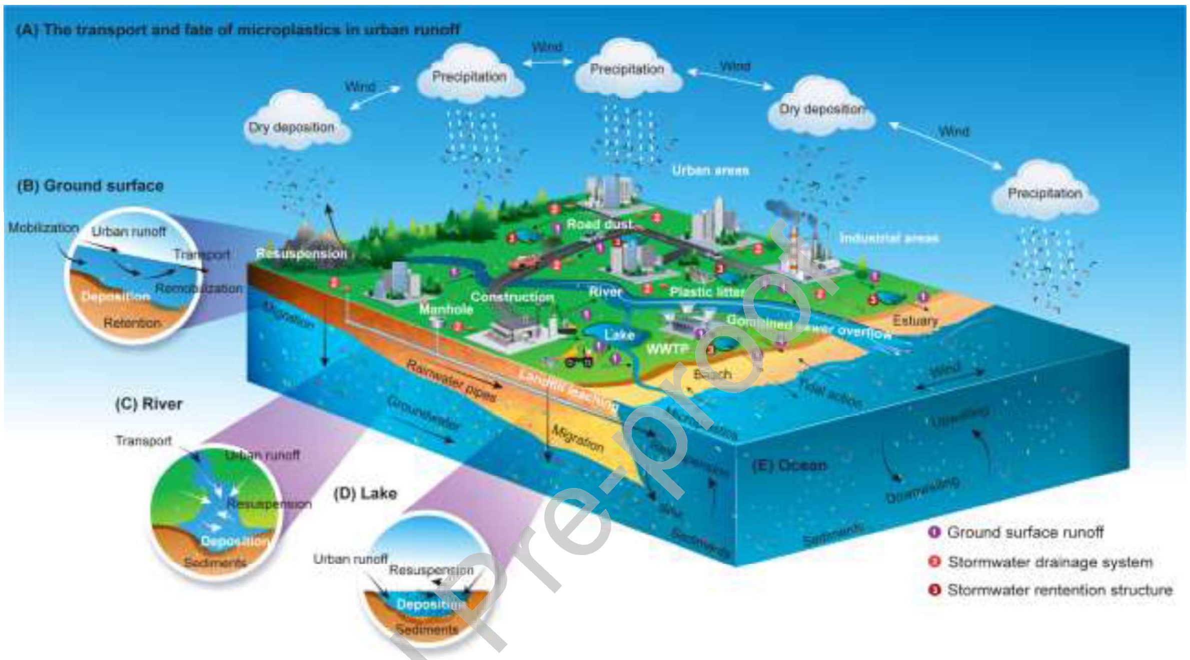


Fig. 4 *Transport pathways, transport and retention mechanisms, and the fate of MPs in urban runoff across different environmental compartments (e.g., terrestrial, freshwater and marine environments).*

3.4 Co-contaminants

In addition to MPs, many other contaminants, including heavy metals and organic pollutants, can also exist in urban runoff (Jarlskog et al. 2021, Muller et al. 2020), which is considered a main non-point source of pollution into receiving waters (Bjorklund et al. 2018, Hajiouni et al. 2022). Due to the physicochemical properties of some MPs, e.g., large specific surface area and strong hydrophobicity, they can have high affinity for hazardous pollutants (Elgarahy et al. 2021, Wang and Wang 2018) and act as a vector to transfer contaminants to different environmental compartments (Gao et al. 2022, Pramanik et al. 2020, Wang et al. 2022). The transport of MPs with adsorbed pollutants in urban runoff can cause serious environmental problems (Ren et al. 2021) and increase the pollution risk of receiving water bodies. Furthermore, some aging processes (Section 3.2) can increase the adsorption capacity of MPs towards other pollutants, thus elevating the risk of co-transport (Hanun et al. 2021, Liu et al. 2022). A recent study explored the effects of polymer type and weathering of low density PE MPs on adsorption of Pb^{2+} in stormwater, revealing significant physio-chemistry changes which increased Pb^{2+} uptake (Aghilinasrollahabadi et al. 2021).

Moreover, MPs can consist of polymers with harmful organic and inorganic additives that are applied as stabilizers, plasticizers and flame retardants (Gonzalez-Ortegon et al. 2022, Teuten et al. 2009). A recent study found that the average concentration of phthalate acid esters detected in urban runoff along the Bushehr coast was 53.57 µg/L, which suggests that plastic additives in MPs pose an environment threat (Hajiouni et al. 2022). Styrene oligomers leached from PS were found to transport from land to the Tokyo Bay via rainfall-induced runoff (Amamiya et al. 2019). N-(1,3-Dimethylbutyl)-N’-phenyl-p-phenylenediamine (6PDD) is widely used as antioxidant agent for automobile tire and it can be oxidized to 6PDD quinone which shows acute toxicity to some species of fishes, e.g., coho salmon (Hiki et al. 2021, Tian et al. 2021). Nowadays, 6PDD quinone has been detected in street dust (Hiki and Yamamoto 2022) and urban runoff (Rauert et al. 2022). However, a wealth of information on the subject exists in published literature and further investigations are needed to better understand the interactions between MPs and other pollutants in urban runoff in different parts of the world.

4 Sinks

There are various sinks for MP in urban runoff. On the one hand, MPs can transfer to water bodies such as streams, rivers, lakes, reservoirs, estuaries and oceans (Lin et al. 2021, Shruti et al. 2021), where they may migrate in the flowing water phase to the ocean (Fang et al. 2021) (**Fig. 4**). On the other hand, they can be deposited in water body sediments or retained in soils, stormwater retention structures or other compartments (Shruti et al. 2021). It is important to review the features of these sinks in order to understand the fate of MPs in urban runoff.

4.1 Rivers

Rivers are not only a sink for MPs from various sources, but also a key conduit of MP transfer from land to ocean (Wang et al. 2020a, Xiong et al. 2019, Xu et al. 2021). Additionally, as interactive and complex dynamic systems, rivers can deposit, retain and remobilize MPs on different spatiotemporal scales (Bai et al. 2022, Eppehimer et al. 2021) (**Fig. 4**).

Discharge of urban runoff affects the dynamics and spatiotemporal distributions of MPs in rivers, with MP abundance showing significant temporal change related to precipitation patterns (Skalska et al. 2020, Wong et al. 2020). Moreover, river sediment acts as the sink for MPs. For example, MPs derived from road marking paints in urban runoff have been detected in sediments of tributaries of the River Thames, UK, at an order of magnitude higher abundance than the water column (Horton et al. 2017a). Furthermore, the input of a large amount of urban runoff can change the abundance distribution of MPs between the water column and sediment, thus increasing the migration of MPs downstream (Xia et al. 2021). For instance, samples of one riverbed before and after flooding suggested that nearly 70% of deposited MPs (43 billion particles) were washed away after a flooding event (Hurley et al. 2018) (**Fig. S1**). There may exist a balance between deposition of newly-introduced MPs during rainfall-induced runoff, and the remobilization of "old" deposited MPs from the sediment as a result of elevated water flow.

A recent study assessed the retention and efflux of MPs in riverbed sediment on a watershed-scale and indicated that MPs entering headwater and mainstream were retained in riverbed sediments at an average rate of 8% and 3% per kilometer, respectively (**Fig. S2**) (Drummond et al. 2022). The variation of retention time of MPs in rivers with different classification levels might be affected by river velocity (Drummond et al.

2022). Another study indicated that the retention factors (% particle retention/km) could affect the total MP load of the river on the Baltic Sea. For example, a retention factor of 2%/km would reduce the total MP load to 3.2% (Schernewski et al. 2021).

4.2 Lakes and ponds

Lakes and ponds in urban areas act as MP sinks (**Fig. 4**), and several studies report that large amounts of MPs are transported to them in municipal wastewater (Laju et al. 2022, Zhao et al. 2022) and urban runoff (Clayer et al. 2021, Grbic et al. 2020, Li et al. 2022). As investigated by Clayer et al. (2021), three major sources of MPs in Lake Mjøsa were attributed to mismanaged waste (24.5 t yr^{-1}), wastewater treatment plant (6.9 t yr^{-1}) and road runoff (5.5 t yr^{-1}), indicating that the contribution of urban runoff to MPs in lakes cannot be ignored. Moreover, Clayer et al. (2021) also calculated the budget of MPs in the Mjøsa catchment and found that large amounts (estimated between 7 and 120 tonnes) of MPs were annually released from land to Lake Mjøsa. Of this mass, only 1 to 9 tonnes remained in the sediment, indicating that most MPs were transported to the ocean through connected water systems.

Another study quantified the concentration of MPs in Lake Ontario, Canada to 0.8 particles/L. The source waters were determined to be urban stormwater runoff (15.4 particles/L), treated wastewater effluent (13.3 particles/L) and agricultural runoff (0.9 particles/L), revealing that urban runoff was a significant contributor of MPs in Lake Ontario (Grbic et al. 2020). Therefore, it is necessary to comprehensively investigate the sources of MPs in lakes, especially urban lakes, to determine the specific contribution of each source in order to model MP fluxes and develop management measures.

Additionally, urban lakes receive MPs not only from urban runoff but also from atmospheric deposition (Allen et al. 2021, Dusaucy et al. 2021, Yang et al. 2022). In this context, rainfall is not only an important driving force for MPs to enter lakes through urban runoff, but also a potent driver accelerating the entry of

MPs into lakes via wet deposition (Klein and Fischer 2019, Xia et al. 2020). However, the specific contribution of MPs from urban runoff and the atmosphere remains unclear. Furthermore, similar to river system, hydrodynamic changes caused by rainfall might lead to the remobilization and resuspension of MPs in lake sediments, which increases MP concentrations in lake water (Xia et al. 2020).

Besides, temporal variation of MP characteristics in lakes is related to temporal variation of those in urban runoff (Lin et al. 2021). For instance, the dominant MPs collected in an Ox- Bow Lake in Yenagoa, Nigeria, were mostly PET (72.63%) in the dry season and PVC (81.5%) in the wet season (Oni et al. 2020). Moreover, the flow of lake water and the renewal speed of lake water are slower than that of rivers, which means that MPs might retain in the lake environment for a long period (Yan et al. 2021, Yang et al. 2022).

4.3 Oceans and beaches

Urban runoff is a major pathway for the dispersion of MPs from the land to marine environments (Castro et al. 2020, da Silva et al. 2022, Exposito et al. 2021).

Firstly, land-based MPs in urban runoff can be transported to the estuary after being transferred to inland water systems, e.g., rivers and lakes, and finally to the ocean (Auta et al. 2017, Piehl et al. 2021) (**Fig. 4**). Meijer et al. (2021) calculated plastic emissions across the processes of mobilization by runoff and wind, the transportation of plastic in rivers, and the transfer of plastic from rivers to the ocean. They reported that rivers account for 80% of the annual global emissions of plastics into the ocean, discharging about 0.8-2.7 million tonnes of plastics into the ocean every year. The discharge of MPs from storm drain systems receiving urban runoff along coastline is also one of the most important routes of MP transfer to the ocean (Balthazar-Silva et al. 2020, Ory et al. 2020). A recent study suggested that more MPs entered the port of Durban, South Africa, from storm water drains than from river sources (Preston-Whyte et al. 2021).

Secondly, MPs accumulated on land during dry periods are seasonally remobilized and "flushed" to coastal waters by rainfall, which can be regarded as a "pulse" input event (Balthazar-Silva et al. 2020, Rios-Mendoza et al. 2021). For example, the discharge of MPs in July and August to the southeastern Baltic Sea accounts for half of the annual input from sewer overflow systems, including urban stormwater runoff (Schernewski et al. 2020). Differences between pulse events and the steady transfer of MPs from land to ocean (Balthazar-Silva et al. 2020) must be taken into account when modelling MP fluxes.

MPs carried in runoff that reaches the coast can either enter the ocean (Jaubet et al. 2021) or deposit on beaches (Kumar and Varghese 2021, Yaranal et al. 2021). Local-scale variability in the distribution of MPs on beaches will depend on tidal cycles and beach hydrodynamics (Balthazar-Silva et al. 2020). Substances leached from MPs have been discovered on beaches. For example, Amamiya et al. (2019) found that styrene oligomers leached from PS were much higher in beach sand (171 times) than in seawater.

In summary, tens of thousands of MPs distributed around the world are collected by runoff every year and finally input into the marine environment. MPs with higher density than seawater or MPs biofouling by microorganisms can settle in seawater and eventually accumulate in marine sediments (Auta et al. 2017). A recent study about MPs in Mediterranean Sea found that mean abundance of MPs in marine sediments (32.4 items/kg) was much higher than in sandy beaches (10.7 items/kg) (Exposito et al. 2021). Additionally, a novel study reported that at least 66.7% of plastics in ocean are difficult to monitor under the observation framework currently adopted worldwide (Isobe and Iwasaki 2022). Therefore, it is very important to investigated the flux of MPs in runoff conveyed to ocean, thus better monitoring the accumulation of marine MPs.

4.4 Soils

Soils can intercept and retain MPs from urban runoff (Nizzetto et al. 2016, Rivers et al. 2021, Yuan et al. 2021). A recent study suggested that road and tire derived MPs could be mobilized by urban runoff and then deposited to soil (Cao et al. 2021). Another study documented that surface runoff affects the accumulation of MPs in urban and industrial soils (Nematollahi et al. 2022). Additionally, a recent study investigated the vertical migration potential of MPs retained in soil media of stormwater control measures (SCM) and found that most of retained MPs were located below the subsurface of SCM, indicating that retained MPs might have the potential to further migrate downwards and pollute groundwater (Koutnik et al. 2022). O'Connor et al. (2019) reported that downward migration of MPs in sandy soil is accelerated by wet-dry cycles. The MPs entering into soils may affect their chemical properties and microbial community (Dissanayake et al. 2022, Palansooriya et al. 2022), thus posing a threat to biodiversity and soil productivity (Hou 2022a, b). Because the MPs and co-contaminants like heavy metals can accumulate in soil over a large time span, this may pose a big challenge for future practitioners who need to remediate the contaminated soil with sustainable means (Hou 2021a, b, Jin et al. 2021, Wang et al. 2021d).

4.5 Stormwater drainage systems and stormwater retention structures

MPs can be retained in stormwater drainage systems and stormwater retention structures due to reduced flow velocity (Boni et al. 2022), thus capturing and intercepting MPs from urban runoff before it enters a receiving water body (Monira et al. 2021b) (**Fig. 4**). A recent study demonstrated that the abundance of MPs in the sediment of rainwater pipelines ranged from 44 to 320 particles/kg (Sang et al. 2021). Liu et al. (2019) reported that, in industrial areas, retained MPs in the sediments of retention ponds can range from 1511 to 127,986 particles/kg, with the hydraulic load positively correlated with the amount of MPs in sediments. The retention phenomenon can be applied in stormwater retention ponds and bioretention systems to reduce downstream MP pollution (Koutnik et al. 2022, Liu et al. 2019, Vijayaraghavan et al. 2021).

There are many kinds of stormwater retention structures available for MP removal, including infiltration trenches (Yano et al. 2021), gross pollutant traps (Lange et al. 2021, Lange et al. 2022), bioretention systems including rain gardens and biofiltration (Boni et al. 2022, Gilbreath et al. 2019, Smyth et al. 2021, Vijayaraghavan et al. 2021), constructed wetlands including stormwater floating treatment wetlands and stormwater retention ponds (Liu et al. 2019, Wang et al. 2020b, Ziajahromi et al. 2020). Other techniques involving coagulation processes, dissolved air flotation and membrane separation can also be applied to remove MPs from urban runoff (Monira et al. 2021a, Monira et al. 2021b).

The properties of MPs, including morphotype and particle size, can affect retention system removal performance. Several studies have demonstrated that fibrous MPs are the most difficult MP morphotype to remove (**Table S2**) (Smyth et al. 2021, Werbowski et al. 2021). Another study documented that the MP concentration in urban runoff was reduced by 84% in the size fraction of 160 μm to 5 mm treated by bioretention systems, indicating effective interception of relatively large MPs in urban runoff (Smyth et al. 2021). Another study reported that the dominant MP size fraction was 190-5000 μm (83.7%) in inlet water samples and 25-100 μm (96.5%) in outlet water samples (**Table S1)** (Ziajahromi et al. 2020), again showing effective removal of larger MPs rather than smaller ones. Among retention systems, bioretention cells and vegetated biofilters have performed best for MP removal, while those that rely on physical separation processes, including gross pollutant traps and sand filters, often perform poorly (**Fig. 5**). Bioretention system is a stormwater treatment facility that combines physical interception, chemical adsorption and bioremediation, and it can improve runoff quality and be integrated into the urban landscape (Vijayaraghavan et al. 2021). Due to its multiple advantages, it has high removal performance of MPs (83-96%) (**Table S2**). Gross pollutant traps are regarded as the main primary treatment level for physically screening or capturing garbage and coarse sediments (Monira et al. 2021b). It can often be combined with other stormwater treatment facilities to improve the efficiency of MP removal and minimize sediment load of combined facilities (Lange et al. 2022). Furthermore, different filter media in the stormwater treatment system have different MP removal performance. A recent study reported that the removal efficiency of 20-100 μm MPs by vegetated biofilters (88%) was significantly better than that by non-vegetated sand filters (47%) (**Table S2**) (Lange et al. 2022). The reasons might be due to the interception of plant roots and the small MPs easier to penetrate deeper into the filter media(Lange et al. 2022, O'Connor et al. 2019). Another novel study found that biochar (Particle size: 5-50 mm) and sand (Particle size: 0.8-1.2 mm) filtration systems had high removal performance (> 90%) for microplastics (90-5000 μm) in stormwater runoff (Pankkonen 2020). The removal mechanism of the former was mainly electrostatic attraction and adsorption, and the latter is mainly interception by small filtration pores and adhesion on the sand surface (Pankkonen 2020). In nature-based solutions (NBS) for urban runoff treatment, settlement, coagulation, and biosorption can occur simultaneously, thus leading to excellent, or even complete, removal rates (Chen et al. 2021, Zhou et al. 2022).

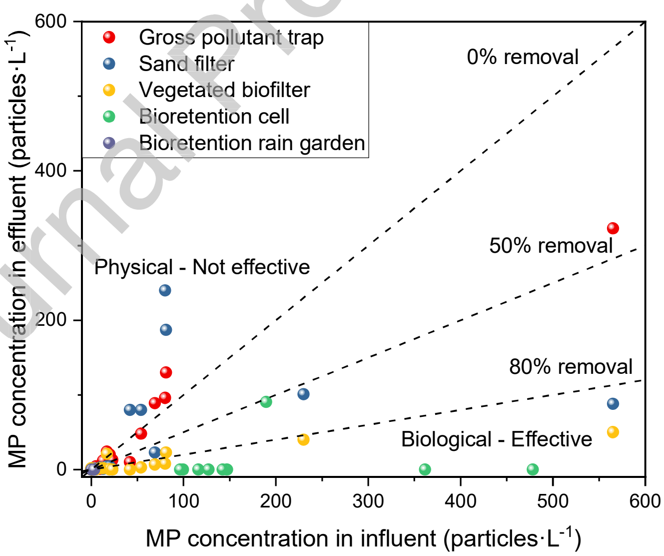


Fig. 5 *Performances of different stormwater retention structures for MP removal. Physical trapping alone may not be sufficient to remove MPs from urban runoff. In contrast, retention structures involving biological processes are more effective for MP removal. Data source: (Gilbreath et al. 2019, Lange et al. 2021, Lange et al. 2022, Smyth et al. 2021, Werbowski et al. 2021).*

5 Implications

With the emergence of urbanization, MP pollution associated with urban runoff has become a global environmental problem. It is necessary to formulate policies, regulations and measures to comprehensively and systematically manage the risks associated with this route of MP migration from land to water bodies.

First of all, source prevention is a crucial step, for which source identification is a prerequisite. Current research on MPs in urban runoff has mainly focused on the concentration, size distribution, morphotype, and polymer identification, rather than quantitative analyses of the contributions of different sources, e.g., road dust and waste plastic dumping. Only when specific sources are identified can the risks be mitigated via source control. However, unlike toxic metals and organic contaminants, whose sources can be precisely identified in environmental investigations by studying stable isotope compositions (Wang et al. 2021b), source identification for MP often relies on basic “fingerprint” features that can be applied in a cost-effective manner. For example, Wang et al. (2021c) proposed that size distribution clusters are important as they form via conditional probability-induced aging processes. Therefore, the cumulative size distribution function of MPs in urban runoff can be regarded as a mixture of various size distribution functions derived from different source materials. The applicability of this method for source identification purposes may be further explored.

In some urban areas, not only are there separate stormwater drainage systems, but also combined sewer systems which can discharge untreated mixed stormwater and wastewater during storm events (Bollmann et al. 2019, Shruti et al. 2021) (**Fig. 4**). This adds to the difficulty for source control. Several studies have reported that urban runoff discharged by combined sewer overflow systems could be considered as an important pathway for MPs in water environments (Chen et al. 2020, Schernewski et al. 2021, Schernewski et al. 2020). More attention must be paid to the impact of separate stormwater drainage systems and combined sewer systems on the discharge of MPs in urban runoff.

Establishing climate-resilient infrastructure to limit urban runoff while simultaneously removing MPs is desirable to prevent MP mobilization and migration. Global climate change adds uncertainty to future storm events (Shaw et al. 2016). Nature-inspired, low-impact development of sponge cities is a promising way to alleviate urban waterlogging and flooding by restoring the natural water cycle and its ecological functions, combined with green and gray infrastructure (Cheng et al. 2022, Qiao et al. 2020, Wang et al. 2021a).

Current stormwater treatment techniques are mainly used to remove total suspended solids, heavy metals, organic matter, and nutrients in stormwater (Al-Ameri et al. 2018, Boni et al. 2022, Valenca et al. 2021). A limited number of works have explored the utilization of stormwater retention structures, e.g., bioretention systems to specifically remove MPs in urban runoff (Gilbreath et al. 2019, Lange et al. 2021). It has been reported that large MPs can be effectively removed by stormwater retention structures (Pankkonen 2020, Smyth et al. 2021), which could efficiently reduce the risk of plastic additives being released into the water environment by MPs (Allan et al. 2022, Do et al. 2022). However, smaller ones, especially nanoplastics (NPs), may escape from these systems and pose the highest environmental risks due to their greater toxicity to aquatic organisms and faster diffusive release rate of plastic additives (Gigault et al. 2021, Wang et al. 2021e). Therefore, stormwater treatment techniques should be redesigned and developed so that they can effectively remove as many MPs with different particle sizes and other pollutants as possible in the urban stormwater runoff simultaneously (Boni et al. 2022). Furthermore, it is essential to evaluate the removal performance of different stormwater retention structures for MPs in urban runoff, and to provide practical guidance for selecting stormwater retention structures to treat urban runoff containing different MP components from diverse land sources (Monira et al. 2021b).

6 Knowledge gaps and future research directions

Migration of MPs in urban runoff from land to freshwater and marine environments further exacerbates MP pollution in aquatic ecosystems. Despite this, investigation on MPs in urban runoff on a global scale is largely inadequate. More detailed and comprehensive research is needed to shed light on the global occurrence and fate of MPs in urban runoff and their impact. Here, we recommend the following research priorities:

1) Quantitative calculation of distribution, concentration and spatiotemporal characteristics of MPs in urban runoff. In particular, experimental and modeling works on a regional or global scale will help us reach a better understanding on the fluxes, environmental behavior and fate of MPs, thus providing a valuable context for policy making and global collaboration.
2) The complexity of MP transport among land, rivers/lakes and oceans should be further taken into account. The “pulse release” of MPs in river or lake sediments related to the intensity of urban runoff processes adds much uncertainty to the flux of MPs to the ocean. Furthermore, other indirect inputs, in particular, atmospheric wet deposition to lakes and ponds should be assessed simultaneously with land-based sources.
3) Fingerprint characteristics of MPs in urban runoff should be further explored to assist in source identification for effective source prevention measures.
4) Standardized methods for sampling, extraction, separation, identification and quantification of MPs in urban runoff should be established. Obtaining high quality, comparable data is a prerequisite for aforementioned quantitative estimation. The different methods currently applied in different studies (**Table 1**) may cause bias that may lead to erroneous conclusions.
5) More holistic research is required to elucidate the underlying mechanisms of dynamic processes in a quantitative manner, such as the role of land use-associated human disturbances on spatial distribution,

effect of rainfall intensity on (re)mobilization of MPs in urban runoff and aging-associated changes in MP migration.

6) Further study on abiotic and biotic reactions for aging, fragmentation and biodegradation of MPs in contamination sources and drainage system to understand formation of MPs in relation to their physical, chemical and biological properties and develop management strategies for MP reduction and prevention.

7) Further exploration of the interaction between MPs and co-contaminants in urban runoff should be conducted. We do not yet know whether metals and organic contaminants are preferentially adsorbed to suspended MP surfaces, or retained in the runoff liquid phase. Basic parameters, such as partition coefficients, should be obtained.

8) More relevant policies, regulations and measures need to be implemented to systematically manage MPs in urban runoff, which requires us to cooperate globally, nationally and regionally, and take action locally.

Declaration of interests

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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