# Riverine microplastics: behaviour, spatio-temporal variability, and recommendations for standardised sampling and monitoring

Karolina Skalska[1*], Annie Ockelford[1], James E. Ebdon[1] & Andrew B. Cundy[2]

[1]Centre for Aquatic Environments, University of Brighton, Brighton, BN2 4GJ, UK.

[2]School of Ocean and Earth Science, National Oceanography Centre (Southampton), University of Southampton, Southampton, SO14 3ZH, UK.

*k.skalska@brighton.ac.uk

***Abstract***

Microplastics (synthetic polymer particles <5 mm in size) are currently of major research interest due to their ubiquity and persistence in the environment, as well as their alleged adverse impact on aquatic biota. Although most research to date has targeted microplastic pollution within the marine environment, riverine pathways deliver up to 80% of plastic debris into the seas and oceans. The transfer mechanisms of microplastics through river systems, however, remain largely understudied. A number of authors have attempted to assess the fate of plastic particles in river systems, often reporting contrasting findings. This is mainly due to the heterogeneity of river systems combined with a lack of standardisation between the sampling protocols adopted. Here, we summarise the current state of knowledge on the riverine transport pathways of plastic debris and examine the typical spatio-temporal patterns in microplastic occurrence in river waters and sediments. Furthermore, we critically evaluate the commonly used sampling techniques and provide guidelines for unified field study design. Lastly, we identify issues that warrant further research and propose recommendations for future studies to improve our understanding of microplastics in the riverine environment. Standardisation of sampling methods will be vital for the development of a more reliable microplastic monitoring strategy and, on a longer timescale, the implementation of appropriate mitigation measures.

**Keywords:** river, plastic pollution, transport, sediment, surface water

## 1. Introduction

Marine plastics have been the subject of considerable international attention, rapidly increasing in profile since the identification of the North Pacific Garbage Patch in 1988 [1]. This has resulted in the development of various high-profile action plans aimed at addressing issues associated with these environmental contaminants (e.g. [2]). Both large 'macro' plastic items (synthetic polymer particles >25 mm in size) and smaller 'micro' plastic items (synthetic polymer particles <5 mm in size) are now widely recognised as an environmental hazard. However, the microplastic sub-group has only recently raised concern due to the presence and persistence of small plastic debris in all environmental media (water, sediments, soil and air), as well as its potential negative effects on aquatic organisms and human health [3–5]. To date, most existing studies have targeted the marine environment, where microplastics can 1) be ingested by a range of aquatic organisms from zooplankton to fin whales (unlike macrolitter, which mainly interacts with larger species) [6–8]; 2) adsorb, and potentially act as vectors for, toxic pollutants [9,10]; 3) harbour harmful microorganisms (e.g. viruses and bacteria) [11,12]; 4) contribute to the spread of invasive species [4,13] and 5) facilitate the spread of antimicrobial resistant genes [14].

The impact of microplastics on riverine systems remains understudied, despite recent estimates suggesting that rivers deliver as much as 80% of the overall load of plastic debris into the global ocean [15]. The annual emission of plastic waste from rivers to the world's oceans has been recently estimated at 0.8 – 2.7 million metric tons [16], further emphasising the role of river channels as major vectors for plastic transfer from land to the marine environment. While microplastics have also been shown to accumulate in standing water bodies (e.g. lakes), understanding their transfer through lotic systems is now crucial, as it may help to close the mass balance of plastic debris present in the environment [17]. Effective mitigation of microplastic pollution will also require a thorough understanding of microplastic transmission processes from terrestrial sources to their sinks (seas and oceans) [18]. Yet, data concerning the environmental behaviour and fate of microplastics in rivers remains scarce, pointing at a fundamental gap in knowledge [19–21].

Research on riverine microplastics is in its infancy and heterogenous, non-standardised methods are used for sample collection and processing, which makes a reliable comparison of studies conducted within different (or even the same) rivers almost impossible [22]. Despite recent efforts to address this issue (e.g. [23]), the most widely applied sampling methods have been developed to primarily investigate microplastics in the marine environment [24]. While

some of these microplastic extraction and identification protocols can be successfully used for certain freshwater compartments (e.g. lakes), care should be taken, as they should not readily be applied to riverine contexts. Sampling river pathways comes with a specific set of challenges. Factors such as the large number of local and diffuse microplastic sources, poor site accessibility, flow fluctuations, high turbidity and diverse channel morphology can lead to a high variation in the concentrations and distribution of microplastics in river networks, as reported by most authors [25–29]. These issues need to be addressed to understand the transport of microplastics through river systems, which in turn will aid implementation of strategies for the reduction of microplastic pollution and help to protect vulnerable riverine and marine ecosystems.

Therefore, the purpose of this paper is to (a) review the extant literature and current state of knowledge regarding the occurrence, behaviour and transfer of microplastics in the riverine environment; (b) critically evaluate the sampling methodologies used to investigate microplastic abundance and dispersal in riverine systems (in both water and sediments); (c) draw attention to the commonly encountered challenges in experimental design and river sampling; (d) identify existing gaps in knowledge and provide recommendations for future research in order to advance our understanding of riverine microplastic pollution.

## 2. Behaviour of microplastics in riverine waters

### 2.1. The influence of intrinsic polymer properties on microplastic transport

To date, a modest number of studies have attempted to explain the behaviour of riverine microplastic inputs and transmission to marine waters, and limited research has targeted the movement of plastic particles [17,30]. Microplastics constitute a highly heterogenous mixture and display different behaviour depending on their physical properties, such as density, shape or size [20], [30]. Plastic density is generally related to its chemical composition, although it can change over time due to natural processes such as ageing and weathering, or biofilm development [18]. Most commercial polymers have a density in the range of 0.85-1.41 $g \cdot cm^{-3}$. For example, polypropylene (PP), polystyrene (PS), polyurethane (PU) and low/high-density polyethylene (LDPE/HDPE) are characterised by a density below 1.00 $g \cdot cm^{-3}$, whereas nylon-6, polyethylene terephthalate (PET) and polyvinyl chloride (PVC) are denser [32]. The low-density materials currently constitute over half of global plastic production, and their manufacturing is predicted to increase over the next decade [33].

The intrinsic properties of plastic particles can determine their ultimate fate in aquatic settings [34]. As confirmed by field observations, the average riverine microplastic pool is mainly

comprised of PE, PP, PVC and PS [25,31,35]. It has long been assumed that since many microplastics remain buoyant in freshwater (1.00 $g \cdot cm^{-3}$), they can be transferred directly from terrestrial sources into the oceans via rivers [34]. Although very high numbers of such low-density polymers are present in rivers, such simplifications should be avoided due to the occurrence of denser plastics in the environment [36]. The density (weight) of microplastics might influence their position in the water column [37,38], i.e. while light debris remains on the water surface, heavier particles can sink directly to the riverbed or be transported as bedload and subsequently accumulate in stagnant zones [39–41]. Indeed, evidence points at the prevalence of low-density polymers on (or directly below) the river water surface, and an increasing abundance of high-density particles towards the bottom sediments (Fig. 1) [42]. Yet, multiple studies confirmed the occurrence of high-density plastics in water columns and conversely, lower-density materials have been detected in sediments [40,43–46].

Once deposited, microplastic particles can infiltrate the riverbed and become buried by the subsequently accumulated bedload [47]. This mechanism is relatively understudied, as most authors collect samples from the sediment surface (e.g. [21,48]; see section 5.3). The accumulation of plastic debris in sediments warrants further research, as new evidence suggests that microplastics can infiltrate streambeds, reaching the biologically important hyporheic zone [49]. Riverbeds may therefore constitute a crucial reservoir for plastic debris [37,50]. However, it remains uncertain whether river sediments are a long-term contaminant sink, as it was recently proposed that previously retained microplastics can be remobilised in higher flows [35,51] (see section 4).

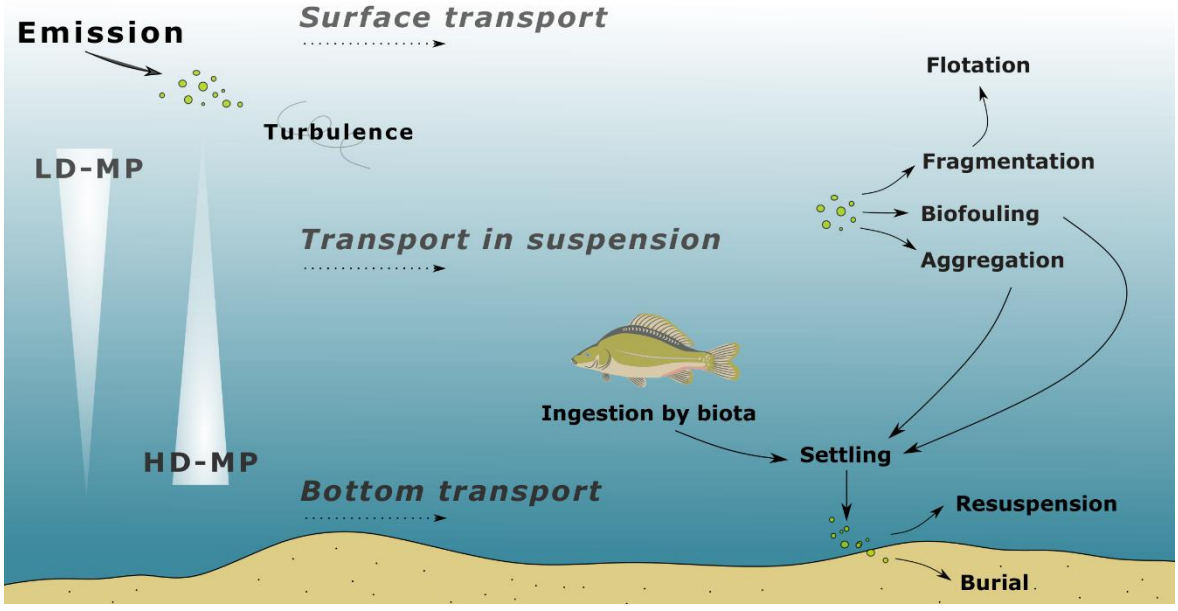


**Fig. 1** Processes affecting the transport of plastic debris through rivers. LD-MP - low-density microplastics (<1.20 g·cm$^{-3}$), HD-MP - high-density microplastics (>1.20 g·cm$^{-3}$). Stock images from https://www.freepik.com/

The settling behaviour of microplastics has recently been explored based on their analogy to allochthonous particulate matter [52]. Similar to microplastics, fine sediment grains generally accumulate in low-energy environments, whereas coarse materials dominate in turbulent, high-energy zones [29,47]. At the same time, it is unclear whether plastic debris might behave similarly to natural sediment grains, which have a much higher density (e.g. 2.65 g·cm$^{-3}$ for quartz) [18]. Several field studies found a correlation between the abundance of certain sediment grain fractions and the quantity of microplastics in river settings. For instance, Corcoran et al. [45] and Tibbetts et al. [40] detected greatest microplastic concentrations in fine sediments [53]. Likewise, Enders et al. [52] described a positive correlation between the concentration of low-density microplastics (<1.20 g·cm$^{-3}$) or microplastics <500 µm and the content of fine (<63 µm) sediment fraction in the Warnow Estuary (Germany). The concentration of high-density microplastics, on the other hand, was explained by the abundance of the local median ($D_{50}$) sediment fraction. Nonetheless, settling of microplastics in the estuarine environment may be influenced by processes such as flocculation and the above findings may not be directly applicable to riverine microplastic studies. Microplastic deposition could also be associated with the shape of debris: while fibres are often denser than freshwater, they tend to remain in suspension longer than spherical beads (pellets) and fragments, which typically settle and accumulate in riverine sediments [38,54,55]. For example, a recent investigation revealed the largest proportion of fibres in sediments enriched in silt and clay, whereas microbeads accumulated in substrate dominated by medium-sized sands (250-500 µm) [29].

In addition to the growing numbers of field studies, microplastic movement has lately been explored using an experimental approach. In line with the settling behaviour of allochthonous organic particles noted in natural settings, a relationship was observed between organic particulate matter and low-density microplastics in an artificial outdoor stream [53]. Particle diameter was determined to be positively related to the depositional velocity of microplastics, which increased in the order of pellets<fibres<fragments. Conversely, another experimental study found large variations in the sedimentation of differently shaped polymers (settling velocity of 0.39 cm·s$^{-1}$ for polyamide fibres vs. rise velocity of 31.40 cm·s$^{-1}$ estimated for expanded polystyrene pellets), suggesting that calculations used for sediment modelling should not be directly used for plastic particles [38]. Concerns have also been raised over the

lack of consistency between environmental microplastics and surrogate synthetic particles used in laboratory experiments. Most laboratory studies involve using homogenous mixtures of microplastics of the same chemical composition, size and shape, which does not correspond to the variety of microplastics found in the field. Moreover, environmental microplastics can exhibit different behaviour due to the influence of natural processes, causing their movement to differ from that of virgin polymers [56]. Although some laboratory simulations reveal clear trends in microplastic behaviour, field-scale findings are often contrasting and suggest it may not be possible to explain the transfer of plastic debris using a single proxy [45,52].

### 2.2. The impact of natural particle interactions on microplastic transport

In field settings, microplastics undergo a range of natural processes, which include degradation, biofouling, aggregation and cycles of ingestion and egestion by biota. In contrast to other freshwater bodies, rivers are highly dynamic and their water residence times are measured in days to weeks, compared to as much as $10^3$ years in lakes [17]. The importance of certain processes (e.g. biofouling) is therefore comparatively low in river waters. Nonetheless, the natural interactions microplastics undergo in riverine environments may be significant in the context of tracking their movement and require further investigation.

Despite displaying certain similarities to fine mineral grains, microplastics are characterised by a lower mechanical strength and undergo fragmentation due to factors such as mechanical stress, UV radiation or biodegradation [57]. While biodegradation of plastic materials is negligible in natural settings, microplastics can break down into smaller items due to turbulence or friction, similar to large plastic litter [58]. In shallow streams, riverine microplastics are exposed to UV light, which weakens their structure and triggers fragmentation into smaller, often submicron particles once the items are subjected to external forces (e.g. by interacting with river flow or sediment) [32]. The magnitude of this process generally increases with decreasing particle size [59,60]. For example, the release of microbeads under 10 µm in size into wastewater can result in a 10-fold increase of microplastic concentrations due to turbulence and the resultant particle cracking [61]. This is in line with field observations, which suggest that the relative abundance of microplastics increases exponentially with their decreasing diameter [62]. Because the adverse effects of microplastics vary according to their size, understanding the fragmentation rates of plastic items will be crucial for modelling future risk associated with the presence of microplastics in the environment [57]. Assessing the degradational patterns of microplastics is also key to

effectively describing their riverine transfer, from the interactions or reactions they undergo (e.g. photo-oxidation due to UV (A/B) exposure or biodegradation; [11,63]) to the physical forces that control their movement (e.g. cracking due to friction; [56], [31]), as the small fragmented particles can be characterised by higher buoyancy in aquatic systems (Fig. 1). Yet, only a handful of studies to date have used SEM (Scanning Electron Microscopy) to investigate the surface characteristics (including degradative features, such as cracking and pock-marks) of collected microplastics [43,64,65].

In contrast, the formation of biofilms on microplastics has received much more attention, mainly due to the ability of synthetic items to harbour invasive species and pathogens [11,13]. Although relatively few studies have described biofilm growth on riverine microplastics, it is likely that the mechanism of 'plastisphere' development in rivers differs from the one prevailing in marine ecosystems as a result of the contrasting conditions in the two environments (UV exposure, salinity, extent of particle weathering, or oxygen content) [66]. The age of plastic particles may influence the occurrence of biofouling in rivers. In some regions, nutrient concentrations in rivers have reduced over the last few decades. As noted by Meng et al. [67], phosphorus content in the River Thames (UK) dropped from 1584 $\mu g \cdot L^{-1}$ in 1998 to 376 $\mu g \cdot L^{-1}$ in 2006 thanks to advances in municipal wastewater treatment, particularly nutrient stripping [68]. This, in turn, may hypothetically reduce the occurrence of biofouling. Indeed, most microplastic studies conducted in UK rivers do not observe biofouling on recovered plastic items [35,48]. While this may be due to decreased nutrient content, the presumably short residence times of microplastics in rivers may also prohibit biofilm growth [67]. Nevertheless, both particle degradation and biofilm growth can have a significant impact on the environmental fate of microplastics, as both processes influence their physical properties (size, shape, density, surface characteristics) [32,63]. Biofouling generally increases microplastic density and facilitates settling, whereas degradation is thought to decrease particle weight, making them more buoyant [69]. This effect might vary between the different shapes of plastic debris: biofouling was found to be more common in fibres, which have a high surface-to-volume ratio and has been proposed as an explanation for the higher abundance of fibres in certain riverbeds [70]. On the other hand, biofilm formation does not appear to occur as frequently on microbeads that are often found to be buoyant in freshwaters [67].

Large quantities of plastic litter are being detected in most rivers around the world, with microplastics locally outnumbering fish larvae [71]. Unsurprisingly, emerging evidence

points at the ingestion of plastic items by freshwater species. Plastic has been found to be consumed both by invertebrates (e.g. *Daphnia magna* [72,73] or Baetidae, Heptageniidae and Hydropsychidae [74]) and vertebrates, such as the common roach (*Rutilus rutilus*) [75], European flounder (*Platichthys flesus*) and European smelt (*Osmerus eperlanus*) [76]. Plastic intake is governed by the particle to mouth size ratio, hence smaller items can generally interact with a wider range of organisms [22,77]. Once ingested by a mobile organism, microplastics can undergo long-range transport through rivers and be excreted far away from the source, often with altered physical properties [31]. Therefore, the cycles of ingestion and egestion by biota constitute an often overlooked, but potentially important influence on the environmental transfer of microplastics.

Lastly, flocculation and aggregation might play a crucial role in microplastic transport through rivers. Research to date has only targeted flocculation of plastic debris in the estuarine or marine environment, as the importance of this process increases with the spike in ionic strength that occurs as microplastics enter saline waters [78]. However, it has been estimated that over 90% of the total volume of sediment can be transported via rivers in flocculated form [79]. By facilitating their sinking during riverine transfer, particle aggregation may impact the mass balance of microplastics. Microplastics interact with other plastic debris (homoaggregation) or natural suspended solids (heteroaggregation) [80]. Heteroaggregation rates depend on the size and density of microplastics, as well as their number concentrations that increase with the decreasing particle size [17]. The shape of debris may also play an important role, as the available surface area influences the probability of a microplastic colliding with other particles. Although aggregation can be easily modelled for spherical items (e.g. using the Smoluchowski particle interaction model), fibres may exhibit different aggregation mechanisms, such as knotting of debris [80]. Moreover, tests conducted in seawater revealed that microplastic aggregation with biogenic particles accelerates following biofilm development [81].

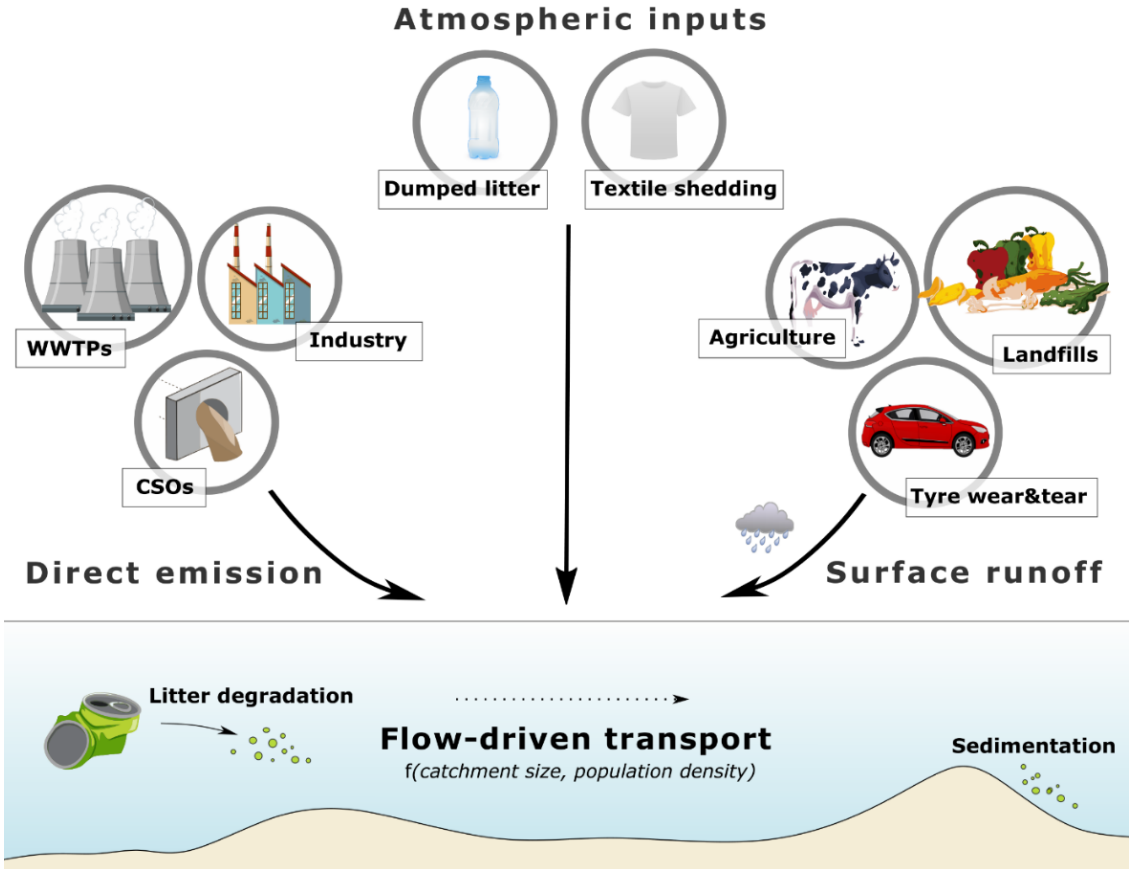
In summary, there is still a lack of understanding of the transport mechanisms microplastics undergo in rivers [82]. Intrinsic particle characteristics can influence their transfer, but research to date shows that they are insufficient to reliably predict the distribution of microplastics in rivers [30]. According to current models, it is physical processes such as particle accumulation and subsequent sedimentation that may ultimately control the fate of microplastics in rivers [80]. However, assessing the behaviour of microplastics is further complicated by factors such as channel morphology, variations in river flow and, most

250 importantly, the highly variable contribution from input sources present across river
251 catchments. The following section therefore aims to summarise the research undertaken on
252 microplastic occurrence and distribution in river systems, and assess the reported spatio-
253 temporal patterns in microplastic abundance in both surface waters and sediments.

254 ## 3. Spatial variability in riverine microplastic concentrations

255 ### 3.1. Influence of point sources

256 Microplastics enter rivers from a variety of point and diffuse sources. While diffuse sources
257 (e.g. littering) are spread over larger areas, point sources include direct inputs from sewers
258 (wastewater effluents), drainage ditches (agricultural runoff) or storm drains (urban runoff)
259 [34]. Over the last decade, great effort has been made to identify such emission points and
260 understand their contribution to microplastic contamination. The vast majority of existing
261 riverine microplastic studies have investigated spatial profiles of plastic debris down river
262 channels. A list of field campaigns conducted to date along with implemented sampling
263 methods, reported microplastic concentrations and the possible pollution sources is compiled
264 in Tables 1 and 2.



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266 **Fig. 2** Main input pathways of microplastics into rivers (WWTP – wastewater treatment plant,
267 CSO – combined sewer overflow). Stock images from https://www.freepik.com/

Overall, microplastic abundance in rivers is a function of catchment population density and size, but it can be influenced by factors such as the level of wastewater treatment and volume of sewage effluent discharged, or the distance from urbanised, industrial and agricultural areas (Fig. 2) [64,83,84]. It is well-documented that microplastic concentrations are often elevated near their emission points, such as large cities, WWTPs (Wastewater Treatment Plants), landfills or plastic manufacturing plants, and typically decrease away from the source [28,31,43]. In particular, the contribution from WWTPs has received considerable attention. Most WWTPs are characterised by relatively high rates of microplastic removal (above 95% in facilities that implement secondary treatment, i.e. sewage filtration and oxidation) [85], with tertiary treatment (e.g. coagulation, reverse osmosis or microfiltration) often removing all plastic particles [86]. Nonetheless, WWTPs are generally inefficient in capturing certain microplastics (particularly microbeads and microfibres) due to their small size and high buoyancy [87]. The overall loads of plastic debris emitted from WWTPs can therefore be substantial (e.g. 500-1000 million microplastics were reportedly discharged per day from three WWTPs in South Carolina (US) [88]), and microplastic concentrations are often elevated near sewage effluent discharge points or CSOs (Combined Sewer Overflows – see section 4) [89–91]. For example, McCormick et al. [92] found higher microplastic concentrations downstream of WWTPs in a small-scale study in Illinois (US), with their abundance shifting from 1.94 $MPs \cdot m^{-3}$ upstream to 17.93 $MPs \cdot m^{-3}$ downstream. Such local increases were later noted in a number of subsequent surveys [93–96]. Kay et al. [97] measured microplastic content in surface waters upstream and downstream of six WWTPs in northern England and found that all of them caused an increase in microplastic abundance in investigated rivers, with the downstream/upstream microplastic concentration ratio reaching as much as 69 (no absolute values reported). At the same time, microplastics were also present upstream of studied WWTPs. This may have been caused by the presence of WWTPs further upstream, but could potentially point at the contribution from the poorly assessed diffuse sources (e.g. atmospheric deposition, soil or road runoff), which could not be excluded.

In contrast, some studies have found no evidence of significant WWTP inputs. Crew et al. [29] investigated microplastic abundance in the sediments and surface waters of St. Lawrence River (Canada) and detected no statistically significant differences between microplastic levels up- and downstream of the ten WWTPs considered in the study, with only a marginal difference in surface water concentrations (an average of 0.12 and 0.16 $MPs \cdot L^{-1}$ up- and

downstream the WWTP outfall). Another study found no correlation between microplastic concentrations and distance (50-1900 m) from WWTPs in surface waters of nine rivers in Illinois (US) [95]. Sewage effluents are often released into areas of high flow, resulting in a rapid dilution of particles and causing the sporadically noted lack of local contamination hotspots [29]. Most authors also observe large inter-site variations downstream of wastewater outlets, due to the different wastewater treatment methods implemented in individual WWTPs [22,95]. Therefore, the population equivalent served by the facility and the utilised treatment methods (e.g. trickling filters vs. activated sludge) should always be considered when assessing microplastic inputs from WWTPs [97].

In addition to WWTPs, large amounts of microplastics are routinely found in proximity to plastic manufacturing plants and other industrial facilities. Microplastics are used in a range of industrial processes as blasting media or raw materials ('virgin pellets') and can be discharged through unregistered or accidental leakage [32]. Several authors noted the presence of such particles in both river waters [71] and sediments [36,98].

Storm drains constitute yet another point source of plastic debris and yield particles originating from abrasion of car tyres and road markings in urban areas [99,100]. The significant contribution from this source was found in some regions of the River Thames (UK) and reflected the importance of vehicle traffic around London [48]. Remarkably, this is reflected by the dominance of tyre wear and tear particles (53%) in relation to other microplastic types present in the oceans, as reported by the Norwegian Environment Agency [101]. A recent study assessed the quantity of tyre particles entering the marine environment via wastewater, storm drains and atmospheric deposition [102]. It was concluded that while tyre particles can originate from all three processes, storm drains represent the most important pathway for tyre particles to enter the seas.

Each point source possesses a unique microplastic fingerprint. Microbeads (used in rinse-off cleaning and cosmetic products, including physical exfoliants) and fibres (from washing of synthetic garments) are generally most abundant near sewage discharge points. Large quantities of polyester fibres have also been observed adjacent to textile factories [44]. Raw plastic pellets (nurdles) are often present close to industrial estates (e.g. plastic manufacturing plants) [25,103], while composite thermoplastic fragments, sometimes containing glass beads for added reflectivity, may be associated with the contribution from storm drains (e.g. chipping of road surface markings) [48]. It is therefore unsurprising that some longitudinal trends in detected microplastic types that relate to locally prevailing contamination sources

are observed along rivers. For example, in the lower Irwell river network (UK), the dominant microplastic type changed from microbeads to fragments returning to microbeads in the Manchester city centre [35]. This reflected the transition from a site with a substantial input of wastewater effluent to a suburban area, and a subsequent return to an urban location with a high contribution from wastewater. The various synthetic particle types recovered from environmental samples can be a useful proxy for determining local microplastic sources and their potential points of entry [36,48].

### 3.2. Influence of diffuse sources

The investigation of longitudinal patterns in microplastic abundance in rivers is complicated by the variable contribution from diffuse (non-point) sources along the catchment, often causing an apparent lack of trends observed even in smaller-scale studies. Whereas point sources often emit characteristic mixtures of microplastics, diffuse sources represent a varied group of debris that undergo different processes and are less well understood [104].

Atmospheric inputs of microplastics have tended to be overlooked, despite estimates suggesting that wind transfer may account for up to 7% of the ocean's plastic pollution [105]. Airborne plastic studies are in their infancy and the contribution of atmospheric microplastics to river pollution has not yet been investigated in any degree of detail [106]. Sources of airborne plastic particles include landfills, waste incineration, industrial emission, particle resuspension, road and tyre wear, sludge and textile shedding [107]. Indeed, microplastics have now been detected in the atmosphere of large cities (e.g. Paris or London) [108,109], as well as in remote locations, such as the Arctic [110] and US national parks [111]. Airborne particles are either directly deposited onto the surface of river waters or enter adjacent streams via runoff from impermeable surfaces [108].

At the same time, the most important contribution comes from the breakdown of larger plastic items [112]. Plastic litter is ubiquitous and highly polluted areas are often found near rivers [58]. For example, a recent study revealed that most Swiss floodplains contain considerable levels of microplastics originating from fragmentation of mesoplastics [113]. Such plastics are emitted to nearby rivers as a result of direct discharge or through wind advection of waste from littered areas/landfill sites. Due to the pronounced turbulence occurring in most rivers, they are later subjected to mechanical stress. This leads to fragmentation of items, which constitutes the main source of secondary (micro)plastics [114]. Despite the importance of this process, weathering of plastic waste has received relatively little attention and a limited number of freshwater studies have accounted for its role in microplastic generation [82], with

only a single field campaign pointing at a positive correlation between macro- and microplastic concentrations (Yangtze River, China) [115].

As mentioned before, increased microplastic concentrations are often noted near urban centres that are usually associated with high wastewater inputs. However, some debris found in such areas originate from surface runoff [116]. Microplastics commonly found in urban zones include particles generated through litter breakdown or tyre wear and tear. Although tyre wear and tear has previously been mentioned in the context of point source microplastic emission (storm drains), these items can also enter rivers with rainwater and surface runoff [104,117].

It is well established that microplastic contamination is usually greater near large cities. Yet, considerable microplastic concentrations are sometimes detected at rural stretches of rivers located near agricultural areas. While lighter debris can be problematic to trap during wastewater treatment, denser microplastics are captured with much higher efficiency and might be incorporated into sludge that is later used as fertiliser [118,119]. Once sludge is applied to land, microplastics can be re-entrained into runoff waters following precipitation, introducing microplastics to adjacent rivers [28,30]. For instance, Kapp and Yeatman [27] sampled surface waters of the Snake River (US) and found the second highest microplastic concentration (9.50 $MPs \cdot m^{-3}$) and a high microfibre content (80%) at a site located near farmland. A range of plastic products are used in farming, including irrigation tape, plastic mulches, hay bale wrap or plastic particles found in soil conditioners [120]. This often leads to chipped fragments being present in riverine systems (e.g. [28]).

Because some microplastics emitted from a single source (e.g. tyre abrasion) can enter rivers through multiple different processes, assessing the contribution from different sources to the contamination found in environmental samples is challenging. Nonetheless, it is evident that diffuse sources of microplastics require much further investigation, and their capacity to conceal trends associated with the presence of point sources along watercourses may be overlooked in many field studies.

### 3.3. The influence of river heterogeneity

Large-scale studies conducted to investigate longitudinal variations in microplastic distribution are often characterised by a poor resolution. Although they sometimes reveal increases in microplastic pollution with respect to proximity to nearby microplastic emission points [26], the inter-site variations in microplastic abundance and characteristics make it challenging to distinguish between local contamination sources and sinks. Such heterogeneity

399 is generally attributed to the influence of physical processes, including turbulence or
400 meteorological events (e.g. floods), on particle dispersal.

401 Rivers are highly heterogenous and numerous factors can underpin the lack of consistently
402 observed spatial patterns (including with respect to proximity to source) in microplastic
403 concentrations (Fig. 3). Overall, microplastics settle in areas of impeded river flow [26]. Such
404 sedimentation of microplastics can happen due to a change in river morphology. Some small-
405 scale variations occur along the river transect, with microplastic concentrations generally
406 being higher along riverbanks, where river flow is reduced relative to the mid-channel [45].
407 Likewise, bedforms such as channel bars influence the travel distance of debris by slowing
408 down the flow, forming important deposition sites for plastic particles [34]. The role of
409 morphology has been highlighted by many authors including Mani et al. [26], who noted a
410 sharp decrease in microplastic concentrations in the water column in the Rhine river stretch
411 characterised by the lowest bed slope (and thus, reduced flow velocity) relative to the rest of
412 the river. Settling can also be associated with the general downstream changes to river
413 characteristics (e.g. the decreasing slope or increasing flow velocity) that occur over large
414 distances. However, few authors have conducted large-scale, high-resolution campaigns that
415 could reveal such general patterns in microplastic distribution. Kapp and Yeatman [27] noted
416 that surface water microplastic concentrations generally increased down-river in the Snake
417 and Lower Columbia Rivers (US), while Mani et al. [26] found higher surface water
418 microplastic levels further down-river in the investigated reach of the Rhine River that
419 spanned over four countries (Switzerland, France, Germany, Netherlands). In contrast,
420 Scherer et al. [98] found decreasing microplastic concentrations in sediments down the River
421 Elbe (Germany). In large-scale studies, microplastic concentrations might be affected by an
422 inflow of water from tributaries: elevated abundances have been found in proximity to
423 channel confluences where an inflow of contaminated waters occurred [25], and conversely,
424 an inflow of less polluted water has been found to have a diluting effect on microplastic
425 concentrations in the investigated channel.

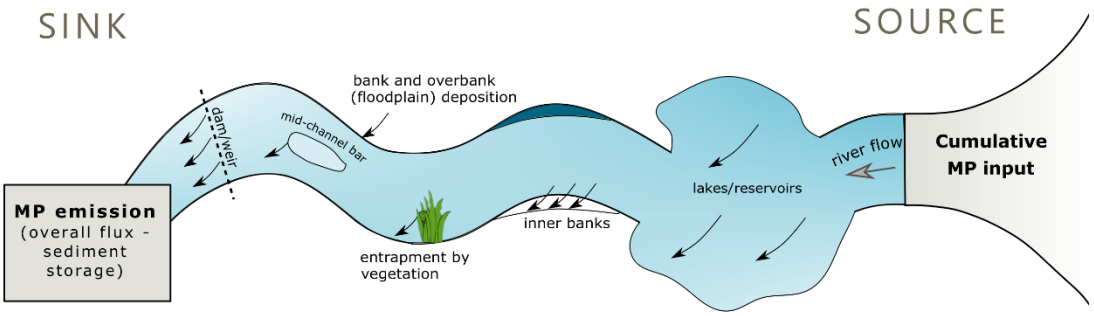



**Fig. 3** Physical factors causing microplastic losses from their source to their sink. Arrows indicate microplastic depositional areas

The presence of anthropogenic barriers, such as dams and weirs, also leads to microplastic deposition and has an effect comparable to natural barriers [39,41]. This was confirmed by Xiong et al. [115], who found that a series of dams located on the upper Yangtze River (China) likely caused substantial retention of microplastics in sediments, forming an important contamination sink. Microplastic concentrations observed downstream of the obstructions were an order of magnitude lower compared to the Three Gorges Reservoir located upstream. Similarly, the presence of reservoirs or lakes along the river channel can cause microplastic accumulation. Watkins et al. [41] reported much higher microplastic concentrations within reservoir water and sediment compared to the values detected upstream. Presumably, microplastics may additionally become trapped and accumulate in in-stream vegetation, which impedes river flow. Large amounts of macroplastics have been found to be retained by hyacinths in the Saigon River (Vietnam) [121]. However, this has not yet been confirmed for microplastics.

In conclusion, rivers often display longitudinal patterns in terms of microplastic concentrations, demonstrating the presence of microplastic sources and sinks along watercourses [31,122]. The heterogeneity of riverine environments means that it is challenging to distinguish between the various contamination inputs [26]. More detailed investigations of individual microplastic sources may allow us to better understand their contributions to catchment-level plastic emission and fluxes, so appropriate mitigation measures can be put in place.

## 4. Temporal changes in riverine microplastic concentrations

In addition to exhibiting spatial patterns, microplastic abundance in river systems can display notable temporal variations, mainly associated with precipitation patterns. Although the extent of such fluctuations depends on factors such as catchment hydrology, local water

management or land-use, several authors have noted increased microplastic concentrations in surface river waters following rainfall events, e.g. in Los Angeles River, US (maxima of 5 $MPs \cdot m^{-3}$ and 153 $MPs \cdot m^{-3}$ noted in the dry and wet season, respectively) [83], Venoge River, Switzerland (150-fold increase in microplastic abundance due to rainfall) [123], Ofanto River, Italy (mean concentrations of 0.9 $MPs \cdot m^{-3}$ and 13 $MPs \cdot m^{-3}$ in the dry and wet season, respectively) [28], or Qiantang River, China (mean of 889 $MPs \cdot m^{-3}$ and 1607 $MPs \cdot m^{-3}$ detected during dry and wet season, respectively) [124]. Because such increases are usually associated with surface runoff, the same can be expected following spring snowmelt, especially in urban and agricultural areas [27,125].

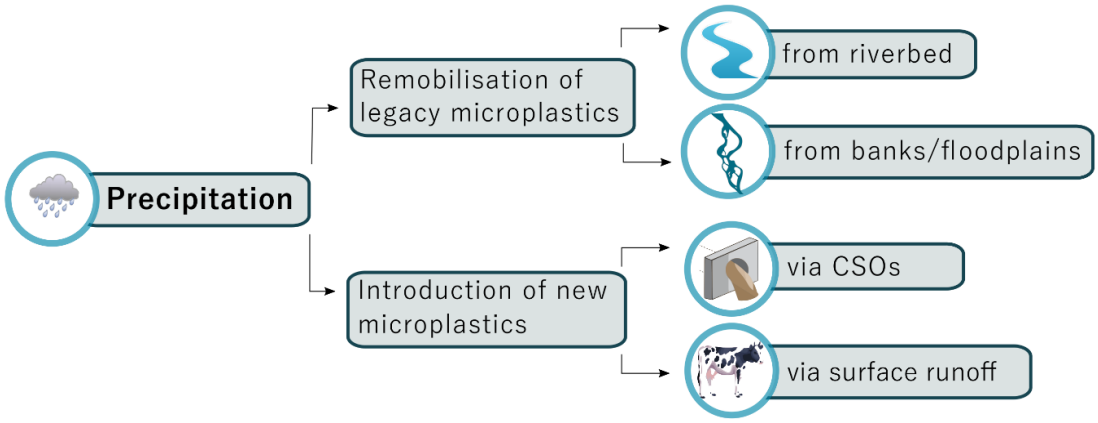
While surface water microplastic concentrations have been widely shown to increase following specific high-intensity rainfall events, transfer of plastic debris through river channels displays pronounced seasonality world-wide. According to a model describing the seasonal variations in river plastic emissions into the global ocean, 74.5% of the total input takes place between May and October, peaking in August (228,800 tonnes) and reaching its low in January (46,200 tonnes) [122]. Although the northern hemisphere (e.g. Europe) experiences the peak microplastic emission earlier in the year (Jan-Apr), the overall figures were largely influenced by the occurrence of the South East Asia Summer monsoon (Nov-Mar). The existence of such seasonal trends in microplastic concentrations is supported by the limited field evidence, e.g. Eo et al. [46] estimated that in the Nakdong River (South Korea), 81% of the annual microplastic load (by weight) can be transported in the three months of the wet season (July-September). A past model additionally suggested that just 10 rivers (Yangtze River, Indus, Yellow River, Hai He, Nile, Ganges/Brahmaputra, Pearl River, Amur, Niger and Mekong) may transfer as much as 94% of the global load of plastic into the seas and oceans [126]. An improved model has now corrected that estimate, revealing that 1000 rivers account for 80% of plastic emissions into the global ocean [16]. Nevertheless, rivers flowing through Asia and Africa carry substantially more plastic litter relative to more economically developed regions due to the mismanagement of waste [100,126]. Better handling of solid waste in these regions could drastically limit the high seasonal input of plastic into the marine environment on a global scale [126]. Given that large plastic items are the main source of secondary microplastics, stopping their spread is a highly effective way of mitigating microplastic pollution [100].

At the same time, evidence suggests that a substantial part of the overall microplastic flux can be transported into the seas and oceans during short-lived high-discharge events. Although a

recent study by Hurley et al. [35] conducted in northwest England (Mersey and Irwell river catchments) reported microplastic concentrations as high as 517,000 MPs·m$^{-2}$ of river sediment, 70% of the overall microplastic burden was eliminated from the catchment after a severe flooding episode. Likewise, Hitchcock [127] observed a 40-fold increase in the abundance of microplastics in the Cooks River estuary (Australia) after a storm event, relative to pre-flood conditions (400 vs. 17,383 MPs·m$^{-3}$ pre- and post-flood, respectively). The elevated microplastic concentrations were positively correlated with the amount of precipitation that occurred over the studied period (5 days). This is in line with the findings of Kudo et al. [128], who noted that microplastic levels increased by an order of magnitude under flood conditions in the Edo River (Japan) and Mintenig et al. [129], who reported a high microplastic concentration (1,494 MPs·m$^{-3}$) in sewage effluent from a Dutch WWTP during a storm event that returned to pre-flood levels (211-711 MPs·m$^{-3}$) two weeks later. Similarly, Treilles et al. [130] observed a 15-fold increase in microplastic concentrations in the Seine River during high-flow compared to base-flow conditions. The same trend has been exposed in numerical models of microplastic transport: Wagner et al. [131] estimated that in the Parthe River (Germany), as much as 90% of annual plastic load could be transferred during just 20% of the year.

While studies have not yet accounted for the mechanism responsible for microplastic increases following high-flow events, authors generally point at the capacity of surface runoff from diffuse sources (e.g. agricultural lands or landfills) and CSOs to introduce microplastics to river waters [18,20,22]. In particular, the role of CSOs in microplastic emission has recently gained much attention. During periods of intense rainfall, stormwater that is collected in the combined sewerage network is released via CSOs that flows to an adjacent river without prior treatment, often resulting in diluted raw (or partially treated) wastewater entering surface water systems [18]. Untreated wastewater often contains large sanitary plastic items and stormwaters have been shown to also contain high levels of microplastics, such as fragments emitted through vehicle tyre abrasion [22,104]. Emissions from CSOs are of particular concern in cities such as London (UK), where outdated sewers are unable to contain high amounts of precipitation and river contamination with foul water following storm events occurs very frequently [132]. For example, van Emmerik et al. [133] measured plastic (>1 cm) transport near the mouth of the River Seine (France) and detected a 10-fold increase in plastic flux during a high-discharge event, with the majority of macroplastics originating from the metropolitan area of Paris. A recent investigation revealed similar CSO

519 inputs yielding high microplastic contamination in the surface water of the River Thames
520 upstream of the City of London (UK) [134]. Such urban inputs are especially pronounced at
521 the 'first flush' of stormwater during flood events due to the mobilisation of contaminants
522 accumulated over dry periods, and decrease over time [135,136]. Conversely, some surveys
523 revealed a decrease in microplastic pollution of surface waters during high-flow events.
524 Barrows et al. [137] found an inverse correlation between microplastic abundance and flow
525 rate in the Gallatin River (US), suggesting a diluting effect of stormwater inputs on surface
526 water microplastic contamination. The investigated catchment was predominantly rural and
527 did not contain CSOs, resulting in further dilution of riverine waters with clean rainwater.
528 This is in line with the findings of Xiong et al. [115], who noted that the large flow of the
529 Yangtze River also caused apparent dilution of microplastic concentrations within surface
530 waters. However, these authors did not account for the presence or absence of CSOs within
531 the catchment.




532

533 **Fig. 4** Mechanisms underpinning the observed increases in surface water microplastic
534 concentrations. Stock images from https://www.freepik.com/

535 Temporal increases in microplastic abundance in surface waters may also result from flushing
536 and re-entrainment of 'legacy' (previously deposited) debris from the riverbed, banks
537 (riparian zones) and floodplains [35,131] (Fig. 4). A number of catchment-scale studies have
538 yielded findings that suggest a significant accumulation of microplastics in riverine
539 sediments, highlighting their role in microplastic trapping. Scherer et al. [98] found
540 microplastic abundance in sediments to be 600,000 times higher than that found in the
541 overlying surface water and Eo et al. [46] found sedimentary microplastic concentrations
542 2,827 times higher compared to the water column. Yet, studies that include longer-term
543 monitoring of microplastic concentrations in river sediments are scarce and yield

contradictory results with regards to the impact of increased flows on microplastic flushing. While Wu et al. [138] found lower microplastic content in Maozhou River (Hong Kong) sediments in the wet season (35-560 MPs·kg$^{-1}$ and 25-360 MPs·kg$^{-1}$ in the dry and wet period, respectively), He et al. [139] noted higher microplastic concentrations in Brisbane River (Australia) sediments in the wet season, conforming to the trend often observed in river waters. According to emerging evidence, much higher flows (i.e. greater shear stresses) may be needed to remobilise microplastics from riverbeds than previously expected [51]. Unreported differences in flow may therefore be responsible for the observed variations in microplastic contamination of sediments and further research is needed to explain the dynamics of microplastic accumulation and remobilisation from riverbeds. This is especially important considering the role of sediments as major microplastic reservoirs [35].

Whether sediments can be a long-term sink of microplastics may also be governed by the characteristics and amount of bed-material overlying microplastics [49]. The role of riverbed stability (i.e. resistance of the bottom substrate to entrainment) in the context of microplastic remobilisation has recently been explored by Ockelford et al. [51]. While fine sediments are re-suspended by a wide range of flows, coarse-grained sediments often form an 'armour layer', i.e. a layer of coarse grains that shields the fine material below from most flows and has been found to substantially limit their exchange across the 'active layer' (i.e. uppermost sediment layer from which particles can be entrained by the flow) [140]. Fine particles accumulated in the sub-surface layer can only be released once the coarse grains have been mobilised [141]. It is therefore highly probable that the water-sediment interface controls the flux of particulate contaminants [51]. The potential of particles to be re-suspended can be further influenced by the presence of biofilms in the sediment ('biostabilisation'), as well as inter-flood duration [142,143]. The impact of sediment stability on microplastics warrants further research, as the occurrence of large-scale microplastic remobilisations can increase the flux of plastic debris to adjacent marine waters and pose a risk to marine ecosystems [51]. This issue may become even more pressing in the light of current climate change scenarios, which predict higher frequency and magnitude flood events in many parts of the world [144,145]. Improved understanding of the mechanism of microplastic flushing may also enhance the management of freshwater microplastic pollution. For example, Liedermann et al. [146] proposed that settling conditions may be artificially enhanced in rivers by constructing structures such as groynes and guiding walls, in order to slow the river flow. Such purposefully built depositional areas imitate the naturally occurring plastic contamination

hotspots observed by several authors [27,35,98] (Fig. 3) and a similar principle is currently being used to remove litter from highly polluted rivers in Asia (via GPTs - Gross Pollutant Traps). Although their efficiency in retaining microplastics has not been assessed, GPTs have been found to successfully trap large plastic items, such as bottle caps and carrier bags [147].

Overall, the mechanism behind apparent storm-related increases in surface water microplastic concentrations remains understudied, but may depend on factors such as the presence or absence of CSOs along the river channel, or sediment characteristics or the shear stress exerted on the streambed by the increased flow. Understanding this process may be vital to reducing riverine microplastic pollution and warrants further research.

## 5. Microplastic sampling in riverine environments

Much progress has been made since the first freshwater microplastic study was published in 2005 [148] in terms of microplastic sampling and analysis methods. Yet, there is currently no standardised protocol for obtaining water column or sediment samples for microplastic analysis and the inter-comparison of studies remains difficult [67]. Most sampling techniques used in freshwater studies were originally developed for marine research [24]. While some of these techniques can be successfully applied to lakes or estuaries, sampling riverine systems poses significant challenges that have not yet received wide attention in the existing literature. As evidenced in section 3.3, rivers are highly heterogenous and many factors can influence the distribution of microplastics. Therefore, tracking the environmental transfer of plastic debris poses a challenge and there is a pressing need for more representative and standardised sampling protocols to be developed. Within this context, this section provides an overview of sampling methods commonly used to investigate microplastics in river waters and sediment, and presents recommendations to encourage good practice in riverine microplastic studies.

### 5.1. Literature search methods

We conducted an extensive literature search in order to review commonly implemented microplastic sampling methods. ISI Web of Science, Scopus and Elsevier ScienceDirect were used to retrieve research articles using the following keywords: ‘microplastics’, ‘freshwater’, ‘riverine’, ‘river’, ‘stream’. Articles containing field data and available online before May 2020 were included in the study, resulting in a total of 47 papers being reviewed.

### 5.2. Microplastic monitoring in surface waters

Microplastic distributions in rivers are usually assessed by sampling surface waters. However, sampling approaches vary between studies. Water collection methods are generally classified as volume-reduced or bulk sampling [149]. Bulk sampling involves collecting water into a

610 container (usually a glass bottle or a plastic container). In volume-reduced sampling, on the
611 other hand, microplastics are filtered out from the water column using a plankton net or a
612 manta trawl [150]. Based on the assumption that most microplastics are buoyant, a trawl is
613 used to collect material from the river surface [151]. The trawl is normally equipped with a
614 flow meter, which allows calculation of the volume of water processed during sampling
615 [152,153]. Both approaches have their limitations and although a uniform water collection
616 method is needed, a suitable sampling protocol should be chosen based on the character and
617 aims of the field study. An overview of methodologies implemented in extant literature is
618 presented in Table 1.

619 **Table 1** A summary of studies carried out to date on the occurrence and distribution of microplastics in river surface waters

| Authors | Location | Study character | Variables considered | Number of sites | Geomorphological unit(s) sampled | Sampling method (area, depth) | Distance/time towed | Volume sampled (L) | Number of replicates per site* | MP lower size cut-off | Potential sources | Reported concentrations (range or average) |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| [94] | Teltow Canal (Germany) | Spatio-temporal | Precipitation, WWTP | 10 | N/R | Bulk water sample (d=0-5 cm) | N/A | Average 12.4 L | 1 | 450 µm | Sewage effluent, surface runoff | **0.01 - 95.80 MPs·$L^{-1}$ |
| [154] | Wuhan rivers (China) | Spatial | Point sources | 16 | N/R | Teflon pump (d=0-20 cm) | N/A | 20 L | 1 | 50 µm | Municipal effluent | 1,660-8,925 MPs·$m^{-3}$ |
| [93] | River Yangtze (China) | Spatial | Point sources, channel confluences, dams | 29 | N/R | Teflon pump (d=1 m) | N/A | 25 L | 2 | 48 µm | Sewage effluent, litter degradation | 1,597-12,611 MPs·$m^{-3}$ |
| [44] | River Saigon and canals (Vietnam) | Spatio-temporal | Precipitation, litter degradation | 6 | N/R | Bulk surface water sample | N/A | 0.3 L | 1 | 2.7 µm | Industrial and sewage effluent, litter degradation | ***172,000-519,000 MPs·$m^{-3}$ |
| | | | | | | Plankton net | 1 min | N/R | 1 | 300 µm | | 10-223 MPs·$m^{-3}$ |
| [108] | River Seine (France) | Spatio-temporal | Point sources, river morphology | 3 | N/R | Manta trawl (d=0-30 cm) | 15 min (2m·$s^{-1}$ fixed speed) | 182 – 200 $m^3$ | N/R | 330 µm | Sewage effluent, atmospheric fallout | 0.28-0.47 MPs·$m^{-3}$ |
| | | | | 4 | Banks, mid-channel | Plankton net (d=0-35 cm) | 1 min | 0.43 - 2.0 $m^3$ | 3 | 80 µm | | 4-108 MPs·$m^{-3}$ |
| [118] | River Raritan (US) | Spatial | WWTPs | 8 | N/R | Plankton net (20 cm diameter, half submerged) | 1 h | 1.3 - 3.5 $m^3$ | 2 | 153 µm | Sewage effluent, litter degradation | 24.0-71.7 MPs·$m^{-3}$ |
| [123] | River Rhone (Switzerland) | Spatio-temporal | Precipitation | 6 | Mid-channel | Manta trawl (0.108 $m^2$) | 15 - 30 min (1.5 m·$s^{-1}$ fixed speed) | 360 $m^3$ | 2-5 | 300 µm | Surface runoff | 7 MPs·$m^{-3}$<br>1.4 mg·$m^{-3}$ |
| [92] | North Shore Channel (US) | Spatial | WWTPs | 8 | N/R | Neuston nets, (a=0.39 and 0.15 $m^2$) | 20 min | N/R | 1 | 333 µm | Sewage effluent | 1.94 -17.93 MPs·$m^{-3}$<br>730,000 – |

| | | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | | | | | | | | | | | | 6,700,000 MPs·km$^{-2}$ |
| [96] | North Shore Channel (US) | Spatial | WWTP | 10 | N/R | Neuston net (a=0.19 m$^2$) | 15-20 min | N/R | 1 | 333 µm | Sewage effluent | 2.4-5.7 MPs·m$^{-3}$ |
| [95] | The North Shore Channel (US) | Spatial | WWTP | 5 | N/R | Neuston net | N/R | N/R | 4 | 333 µm | Sewage effluent | 1.67-10.36 MPs·m$^{-3}$ |
| [83] | Los Angeles and San Gabriel rivers (US) | Spatio-temporal | Precipitation, water depth | 3 | River/bank interface, mid-channel | Manta trawl (a=0.14 m$^2$, mid-channel surface) | 30 s-15 min | N/R | 3 | 333 µm | Litter | <1 – 153 MPs·m$^{-3}$/<1-81 g·m$^{-3}$ wet season 0-5 MPs·m$^{-3}$/<1 g·m$^{-3}$ dry season |
| | | | | | | Handnets (a=0.08 m$^2$, bank surface) | N/R | N/R | 3 | 500 µm 800 µm | | 10-271 MPs·m$^{-3}$/<1-40 g·m$^{-3}$ wet season 0-22 MPs·m$^{-3}$/0-1 g·m$^{-3}$ dry season |
| | | | | | | Streambed sampler (mid-depth/bottom) | N/R | N/R | 3 | 333 µm | | <1-123 MPs·m$^{-3}$/<1-123 g·m$^{-3}$ wet season 0-<1 MPs·m$^{-3}$/0-<1 g·m$^{-3}$ dry season |
| [71] | River Danube (Austria) | Spatio-temporal | Point sources | 3-4 | N/R | Drift net (50 cm diameter) | 30 min | N/R | 2/3 | 500 µm | Plastic production | 317 MPs·1000 m$^{-3}$ |
| [26] | River Rhine (Netherlands, Switzerland) | Spatial | Point sources, river morphology | 11 | Banks, mid-channel | Manta trawl (a=0.11 m$^2$) | 15 min | 60-250 m$^3$ | 3 | 300 µm | Industrial and sewage effluent | 4,960 MPs·1000 m$^{-3}$ 892,777 MPs·km$^{-2}$ |
| [155] | Qinghai Lake tributaries (China) | Spatial | N/R | 4 | N/R | Manta trawl (a=0.5 m$^2$) | 10 – 20 min | N/R | 1 | 112 µm | N/R | 3,000-31,000 MPs·km$^{-2}$ |

| | | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| [39] | Three Gorges Reservoir and tributaries (China) | Spatial | Distance to a dam | 5 | N/R | Manta trawl (a=0.5 $m^2$) | 400 m | 500 mL | 1 | 112 µm | Municipal effluent | 192,000-13,600,000 MPs·$km^{-2}$ |
| [156] | River Antuã (Portugal) | Spatio-temporal | Precipitation, point sources | 3 | N/R | Motor water pump | 5 min for surface and bottom water | 1.2 $m^3$ | 1 | 55 µm | Municipal effluent | 58–193 MPs·$m^{-3}$/5.0-8.3 mg·$m^{-3}$ (March) 71–1,265 MPs·$m^{-3}$/5.8–51.7 mg·$m^{-3}$ (October) |
| [157] | Tibet Plateau rivers (China) | Spatial | Point sources | 6 | N/R | Large flow sampler | N/R | 30 L | 3 | 45 µm | Municipal effluent | 483-967 MPs·$m^{-3}$ |
| [115] | River Yangtze (China) | Spatial | Point sources | 15 | N/R | AVANI trawl (a=0.11 $m^3$) | 15-30 min | N/R | 1 | 333 µm | Litter degradation | $1.95\times10^5$-$9.00\times10^5$ MPs·$km^{-2}$ |
| [158] | River Qin (China) | Spatial | Point sources | 12 | N/R | Teflon pump | N/A | 20 L | 3 | 25 µm | Aquaculture, sewage effluent | 16.67-611.11 MPs·$m^{-3}$ |
| | | | | 7 | N/R | Plankton net (d=0-20 cm) | 30 min | 7800–160500 L | 1 | 75 µm 300 µm | | 0.1-5.6 MPs·$m^{-3}$ 0.1-4.6 MPs·$m^{-3}$ |
| | | | | 3 | N/R | Trawl net | 10 min | N/R | 1 | 300 µm 75 µm | | N/R |
| [137] | River Gallatin (US) | Spatio-temporal | River discharge, channel confluences, point sources | 72 | N/R | Bulk water sample | Average 1.3 L | 1 L | 2 | 100 µm | Recreational activity | 1-67,500 MPs·$m^{-3}$ |
| [159] | Amsterdam canal (Netherlands) | Spatial | WWTPs | 6 | N/R | Bulk grab water sample | N/A | 2 L | N/R | 10 µm | Sewage effluent | 48–187 MPs·$L^{-1}$ |
| | Rhine and Meuse rivers | | | 3 | N/R | Continuous centrifugation | N/R | N/R | 2 | 10 µm | | 1,400–4,900 MPs·$kg^{-1}$ d.w. |

| | | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | (Netherlands, Germany) | | | | | system (SPM samples) | | | | | | |
| [116] | Estuarine tributaries within the Chesapeake Bay (US) | Spatio-temporal | Precipitation, point sources | 4 | N/R | Manta trawl (width=70 cm, d=0-15 cm) | 1-2 km | N/R | 3 | 330 µm | Surface runoff from urban areas | <1->560 g·km$^{-2}$ |
| [28] | River Ofanto (Italy) | Temporal | Precipitation, river level and flow | 1 | Mid-channel | Plankton net (a=0.30 m$^2$, d=0-45 cm) | 30 min | N/R | 6 | 333 µm | Surface runoff from agricultural lands | 0.9-13.0 MPs·m$^{-3}$ |
| [27] | Snake and Lower Columbia rivers (US) | Spatial | River flow, point sources | 26 | 3-4 m from the bank | Bulk water sample | N/A | 1.85L | 1 | 100 µm | Surface runoff from agricultural lands | 0-5.41 MPs·L$^{-1}$ |
| | | | | | Bank/mid-channel | Circular plankton net (d=0-25 cm) | Average 72 s | 3,207 L | 1 | 100 µm | | 0-0.014 MPs·L$^{-1}$<br>0-13.7 MPs·m$^{-3}$ |
| [125] | Great Lakes tributaries (US) | Spatio-temporal | Point sources, river flow | 29 | N/R | Neuston net (a=0.40 m$^2$, d=20-35 cm) | 5-82 min | N/R | 1 | 333 µm | Litter degradation and other diffuse sources | 0.05−32.00 MPs·m$^{-3}$ |
| [97] | Northern England rivers (UK) | Spatial | WWTP | 6 | N/R | Plankton net (a=0.06 m$^2$) | 15 min | N/R | 3-5 | 300 µm | Sewage effluent, litter degradation | N/R |
| [29] | River St. Lawrence (Canada) | Spatial | Point sources | 10 | N/R | Mesh filtration (d=0-5 cm) | N/A | 100 L | 3 | 100 µm | Urban land runoff | 0.12 MPs·L$^{-1}$ upstream WWTP<br>0.16 MPs·L$^{-1}$ downstream WWTP |
| [129] | Meuse and Dommel rivers (Netherlands) | Spatio-temporal | WWTP, river discharge | 25 | N/R | Centrifugal water pump over stacked sieves (20-300 µm; d=0-5 cm) | N/A | 1.3-8 m$^3$ | 1 | 20 µm | Urban runoff | 67-11,532 MPs·m$^{-3}$ |
| [131] | River Parthe | Spatio- | WWTP, land | 2 | River thalweg | Stationary | 24 h | N/R | 10 | 500 | Urban | 74·10$^{-3}$ |

| | | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | (Germany) | temporal | use, river flow | | | floating drift nets (a=0.90 $m^2$; d=0-20 cm) | | | | µm | runoff | $MPs \cdot m^{-3}$ |
| [160] | River Tamsui (Taiwan) | Spatio-temporal | Point sources, flow, precipitation | 4 | Bank/mid-channel | Manta trawl (a=0.045 $m^2$) | 5 min | 13.7-61.9 $m^3$ | 3-5 | 300 µm | Industrial activity | 2.8-83.7 $MPs \cdot m^{-3}$ |
| [124] | River Qiantang (China) | Spatio-temporal | Point sources, precipitation | 12 | N/R | Steel bucket, surface water filtered through two sieves (45 µm, 5 mm) | N/A | 20 L | 1-2 | 45 µm | WWTPs, textile production, urban runoff | 889 $MPs \cdot m^{-3}$ (dry season) 1,607 $MPs \cdot m^{-3}$ (wet season) |
| [46] | River Nakdong (South Korea) | Spatio-temporal | WWTP, precipitation | 9 | Bank/mid-channel | Steel beaker (d=0-20 cm) Submersible pump (d=1 m) | N/A | 100 L | 1 | 20 µm | WWTP | 293- 4,760 $MPs \cdot m^{-3}$ |
| [161] | River Wei (China) | Spatial | Point sources, flow | 15 | Bank/mid-channel | Bulk water sample | N/A | 30 L | 3 | 75 µm | Surface runoff from agricultural lands | 3.7-10.7 $MPs \cdot L^{-1}$ |
| [98] | River Elbe (Germany) | Spatial | Point sources, flow | 10 | Harbour entrances/ fine sediment accumulation zones (banks) | Apstein plankton net (a=0.022 $m^2$) | 5-10 min | 3.2–32.7 $m^3$ | 1 | 150 µm | Sewage effluents, industrial emission | 5.57 ± 4.33 $MPs \cdot m^{-3}$ |
| [134] | River Thames (UK) | Spatio-temporal | Point sources, flow, river depth, tidal activity | 2 | 10-15 m from the bank | Ichthyoplankton net (a=0.90 $m^2$) | 5 min | N/R | 5-6 | 32 µm | CSO inputs, urban runoff | 24.8 $MPs \cdot m^{-3}$ (Putney) 14.2 $MPs \cdot m^{-3}$ (Greenwich) |

*Where no replicates were mentioned in the study, it was assumed that a single sample per site was collected

N/R – not reported, N/A – not applicable, WWTP – wastewater treatment plant, d – depth, a – area, MPs – microplastics (count)

Manta nets typically have a mesh of 300 or 330 µm, with the cut-off point corresponding to the lower size threshold of microplastics proposed by the National Oceanic and Atmospheric Administration of USA (NOAA) [149,162]. However, other mesh sizes such as 112 µm [39], 153 µm [118] or 500 µm [71] have been used. The mesh size controls the number and size of collected microplastics. For instance, Dris et al. [163] recovered 30 times more microplastics using a plankton net with a mesh of 100 µm relative to a manta net (330 µm), and a recent study by Lindeque et al. [164] revealed similar differences: mesh nets of 333 µm and 500 µm retained 2.5 and 10 times less microplastics compared to a 100 µm net, respectively. Even more strikingly, Lozano and Mouat [165] reported a 100,000-fold increase in microplastic abundance when using 80 µm mesh compared to a mesh of 450 µm. Manta trawls are also ineffective in trapping synthetic fibres, with an 80 µm mesh being able to filter as many as 250 times more filaments [108]. This suggests that many studies could be underestimating the number of microplastics present in river systems, and the smallest particles that are of particular concern due to their adverse impact on aquatic biota and human health might be overlooked [7,164]. Indeed, several authors found the majority of detected microplastics to be under 100 µm in size (e.g. 67.1% of particles in the Meuse river basin (Netherlands)) [129]) and only one study to date reported no difference in the abundance, particle size and type of microplastics retained by plankton nets with the two contrasting pore sizes (75 µm and 300 µm) [158]. However, the latter finding likely resulted from the local characteristic of microplastic pollution, as most microplastics captured in the study were in the 1-5 mm size range.

On the other hand, smaller mesh nets have been shown to quickly become clogged with suspended particles and may therefore be deployed for a shorter amount of time depending on water turbidity [27]. An extension in sampling duration from one to three minutes has been reported to greatly reduce intra-site variations [37]. Using nets with a smaller mesh can hence lead to a poorer representativeness of collected data. In an experimental study by Dris et al. [166], filtration using an 80 µm mesh became ineffective after a minute of sampling (i.e. when 8 $m^3$ water passed through, with a background total suspended solids concentration of 10 $mg \cdot L^{-1}$). On average, plankton nets (80 µm mesh) allow to sample 2 $m^3$ of surface water, whereas manta trawls (300-333 µm mesh) can filter as much as 200 $m^3$. This effect is especially pronounced in rivers containing high levels of organic matter and may vary on a seasonal basis due to the occurrence of algal blooms, or depending on river traffic [112]. Mesh clogging can generally be overcome by taking repeat samples or using filter cascades of

655 different mesh sizes that fractionate suspended particles, increasing the volume of water that
656 can be processed [37]. Nonetheless, the use of such samplers is associated with the possibility
657 of irreversible retention of particles that increases with the number of mesh filters used, and
658 the use of cascades can be associated with a loss of microplastics that should be assessed prior
659 to sampling [167].

660 Another challenge associated with using trawl methods has been highlighted by Prata et al.
661 [151]. The trawl protocol was originally developed for seawater, which has a density of
662 around 1.03 $g \cdot cm^{-3}$. Meanwhile, less microplastics remain buoyant in freshwater that has a
663 density of 1.00 $g \cdot cm^{-3}$ [151]. Although some studies report a general prevalence of
664 microplastics on the water surface (e.g. 3 times higher microplastic concentrations in surface
665 waters relative to benthic water observed by Eo et al. [46]), river morphology and
666 hydrodynamics can influence the position of microplastics in the water column [26]. Higher-
667 density polymers are generally expected to prevail near the river bottom and conversely, low-
668 density materials accumulate near the water surface [42]. At the same time, processes such as
669 aggregation and biofouling further increase the density of plastic particles, causing them to
670 settle [82]. A recent study revealed non-uniform patterns of microplastic contamination along
671 depth profiles in Hillsborough River (US) that were highly dependent on the dominant river
672 hydrodynamic profiles [168]. Surface water samples are usually taken from depths up to 25-
673 30 cm from the free surface [146] (see Table 1), despite field evidence suggesting it may be
674 appropriate to also collect water samples from deeper in the water column [151]. This has so
675 far only been done in a few studies, where direct filtration of water with submersible Teflon
676 pumps and collection of bulk samples have been successfully employed (e.g. [93]).
677 Nevertheless, the investigation of microplastics in surface waters with mesh nets is easier,
678 more accessible and therefore the most common.

679 A much smaller number of studies have assessed microplastic abundance in surface waters
680 using bulk water samples. Volumes of samples collected vary, with most authors taking up to
681 25 L of water. Although it is appreciated that collecting larger amounts of water improves the
682 representativeness of data, a representative sample volume has not yet been defined for
683 investigating microplastics in river systems [151]. Due to the lack of a particle size cut-off
684 point, bulk sampling can have an advantage over a standard manta net. Experimental studies
685 suggest that the number of particles captured increases in the order of manta net (330
686 µm)<bulk water<hand net (50 µm) [169]. However, it has been estimated that large volumes
687 of water (approx. 100 L) are required to obtain reliable results, whereas most freshwater

studies include obtaining much smaller volumes of water. One of the benefits of collecting bulk water samples is the potential for reducing contamination, as nylon nets can potentially introduce microplastics to the obtained samples [151]. While the use of clean glass bottles helps to overcome contamination, it limits the volume of sample processed. In contrast, using pumping systems can be a more efficient method of collecting bulk samples where large amounts of water are needed, and such equipment is often employed on research vessels. On the other hand, pumping systems are often plastic (e.g. Teflon pumps) and their contribution to microplastic contamination can be hard to assess [170]. Shear stress generated by pumping can also lead to a breakdown of microplastics into smaller particles, potentially skewing particle size distribution and yielding false results [61]. Due to the ease of sample handling and processing, volume-reduced sampling is generally recommended when many sampling sites are considered in the field study.

Microplastic studies often face criticism due to the varying units concentrations are reported in [171]. Microplastic levels are generally expressed in particle count per sample volume where authors utilise the bulk sampling approach. In contrast, those who collect volume-reduced samples state concentrations in particle count per area sampled (calculated by multiplying the trawl width by distance towed) [67]. It is therefore recommended that where possible, results obtained using the latter method should be re-calculated and additionally reported per volume of water to enable a better inter-comparison of studies.

Finally, site accessibility should be considered prior to sampling, as it may determine which water collection method will be the most suitable. Accessibility of a location is dependent on the size and depth of the river channel, slope of the riverbank, meteorological conditions and other factors (e.g. situation on private vs. public land). In larger rivers, using a vessel is necessary to take measurements of microplastic abundance across the entire channel cross-section. At the same time, boats can generate turbulence and disturb surface water microplastics [158]. The net should therefore be towed by the side of the vessel with a suitable gap (2-4 m) [26]. Alternatively, equipment can also be lowered from bridges or cranes [83]. In shallow rivers, wading into the river to collect samples is often reported (e.g. [35,45]). Manually collecting samples from the cross-section may be impossible where deep erosion occurs at outside banks in meandering rivers. Water levels may also fluctuate on a seasonal basis, restricting site access during extreme low-flow or high-flow periods. Collecting samples while standing on a riverbank, on the other hand, provides limited access to the river channel, meaning samples are often constrained to near bank locations.

In conclusion, employing manta trawls remains the most common microplastic collection method in riverine studies. In addition to the ease of sample collection, manta nets allow the filtering of large volumes of water, thus ensuring the representativeness of data [151]. The popularity of the method also enables some degree of standardisation and inter-comparison between studies. Using the standard mesh size may be appropriate for monitoring microplastic concentrations in river waters. However, manta nets fail to capture the smallest (and possibly the most abundant and harmful) microplastic fractions and we recommend that a smaller mesh net is incorporated in studies that involve risk assessment of plastic debris.

### 5.3. Microplastic monitoring in river sediments

Riverine microplastic studies which incorporate sediment sampling are less common than those that involve the collection of samples from the water column (Tables 1 and 2). Water samples are generally much easier to obtain, while sediment extraction usually involves using specialised equipment, adding cost to the study [151]. Most authors use grab samplers or stainless-steel shovels (manual extraction) to collect sediment. Sediment cores, although commonly used in other freshwater compartments (lakes and reservoirs), are not typically used in riverine microplastic studies that typically only monitor the microplastic contamination of surface sediments. Choosing between manual extraction of sediment and employing a specialised sampler is conditioned by site accessibility and riverbed characteristics. The manual extraction of sediment requires wading out into the river, restricting its use to shallow streams. In contrast, instruments such as Ponar or Van Veen grab samplers can be lowered from a vessel, bridges or riverbank and can be used in larger river systems year-round. While sediment grabs perform well in sampling fine material, they are not suitable for retrieving bedrock or samples in coarse-grained alluvial rivers. For instance, Xiong et al. [115] could not extract sediments from some sampling sites due to consolidated substrate present in certain areas of the Yangtze River (China). Where possible, the river bottom can be visually scanned prior to deploying a sediment grab to avoid large boulders [45].

Although manual collection allows extraction of coarse material, it may cause a loss of fines and lead to an underestimation of microplastic abundance in the samples. For example, Petts [172] reported a 5-fold increase in the concentration of fines (<2 mm) using freeze-sampling compared to bulk sampling in a gravel stream. On the other hand, both freeze and bulk sampling exposed the same spatial pattern in fine particle content. The wash-out of fines can also be limited by using steel cylinders that shields the collection area from the river flow

754 [173]. Despite issues associated with its successful implementation, manual sampling
755 represents an accessible, cost-effective method for microplastic monitoring that can produce
756 reproducible data if utilised on a global scale. Nevertheless, grab samplers facilitate a good
757 representation of the entire grain size distribution of the sediment bed and generate more
758 reliable results overall.

759 **Table 2** A summary of studies carried out to date on the occurrence and distribution of microplastics (MP) in river sediments

| Author | Location | Sampling method | Study character | Variables considered | Number of sites | Geomorphological unit(s) sampled | Amount of sediment collected | Depth | Number of replicates per site* | MP size cut-off | Potential sources | Reported concentrations (range or average) |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| [139] | River Brisbane (Australia) | Ponar grab | Spatio-temporal | Point sources | 22 | Mid-channel | N/R | 3 cm | 1 | <1 mm | No clear trends due to flow conditions | 0.18-129.20 $mg \cdot kg^{-1}$ 10-520 $MPs \cdot kg^{-1}$ |
| [45] | River Thames (Canada) | Ponar grab (0.023 $m^2$) | Spatial | Land use, river morphology, GSD, OM | 34 | Mid-channel, inner or outer bend | N/R | N/R | 1 | 63 µm-5.6 mm | Paints, textiles | 6-2,444 $MPs \cdot kg^{-1}$ d.w. |
| [48] | River Thames (UK) | Manual (scoop) | Spatial | Point sources | 4 | N/R | 250 g | 10 cm | 4 | 1-4 mm | Road markings, sewage effluent | 66 $MPs \cdot 100g^{-1}$ |
| [47] | River Bloukrans (South Africa) | Manual | Spatio-temporal | River flow, depth, OM, GSD, channel width, sources | 32 | Random | 2 kg | 5 cm | 1 | 63 µm | Litter degradation, CSOs | 6.3±4.3 $MPs \cdot kg^{-1}$ d.w. (summer) 160.1±139.5 $MPs \cdot kg^{-1}$ d.w. (winter) |
| [115] | River Yangtze (China) | Ponar grab | Spatial | Presence of mesoplastics, flow | 8 | N/R | N/R | N/R | 1 | N/R | Degradation of mesoplastics | 7-66 $MPs \cdot kg^{-1}$ |
| [158] | River Qin (China) | Grab dredge | Spatial | GSD, point sources | 12 | Mid-channel, both bends | N/R | 5 cm | 3 | <0.075 mm | Aquaculture, sewage effluent | 0-97 $MPs \cdot kg^{-1}$ d.w. |
| [43] | River Beijiang (China) | Manual (shovel; 0.04 $m^2$) | Spatial | N/R | 8 | N/R | N/R | 2 cm | 3 | <5 mm | Litter degradation | 178-554 $MPs \cdot kg^{-1}$ d.w. |
| [25] | River Rhine (Germany) | Manual (spoon, 0.003 $m^2$) | Spatial | N/R | 8 | Random | 3-4 kg | 2-3 cm | 35−40 | 63 µm | No clear trends | 228-3,763 $MPs \cdot kg^{-1}$ d.w. 21.8-932 $mg \cdot kg^{-1}$ |
| [174] | River | Manual | Spatial | N/R | 7 | Banks | 500 g | 5 cm | 3 | <100 µm | Spills, litter | 178-544 MPs·1 |

| | | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | Shanghai (China) | (shovel, 0.25 $m^2$) | | | | | | | | | degradation | 00g d.w. |
| [159] | Amsterdam canals (Netherlands) | Van Veen grab | Spatial | N/R | 6 | N/R | 1 L | N/R | >2 | 10 µm | Municipal effluent | <68-10,500 MPs·$kg^{-1}$ |
| [175] | River Xiangjiang (China) | Manual (shovel) | Spatial | N/R | 12 | N/R | 1 kg | 5 cm | 1 | <0.5 mm | No clear trends | 27-866 MPs·$kg^{-1}$ |
| [93] | River Yangtze (China) | Van Veen grab (0.25 $m^2$) | Spatial | Point sources, channel confluences, dams | 29 | Mid-channel | 1 L | N/R | 2 | 48 µm | Sewage and industrial effluent | 25-300 MPs·$kg^{-1}$ |
| [36] | River St. Lawrence (Canada) | Ponar grab (0.023 $m^2$) Peterson grab (0.093 $cm^2$) | Spatial | N/R | 10 | N/R | N/R | 10 cm (Ponar grab) 10-15 cm (Peterson grab) | 5 | 500 µm | Municipal/industrial effluent | 13,759 MPs·$m^{-2}$ |
| [20] | River Elbe, Mosel, Neckar, Rhine (Germany) | N/R | N/R | N/R | N/R | N/R | N/R | N/R | 1 | <5 mm | N/R | 34-64 MP·$kg^{-1}$ d.w. |
| [156] | River Antuã (Portugal) | Van Veen grab | Spatio-temporal | River flow | 3 | Bank | 0.012 $m^3$ | 12 cm | 2 | 55 µm | Sewage effluent, litter degradation | 100–629 MPs·$kg^{-1}$ /13.5–52.7 mg·$kg^{-1}$ (March) 18– 514 MPs·$kg^{-1}$ /2.6–71.4 mg·$kg^{-1}$ (October) |
| [157] | Tibet Plateau rivers (China) | Manual (shovel. 0.04 $m^2$) | Spatial | N/R | 6 | Bank | 200 g | 2 cm | 1 | N/R | Everyday activity of residents (sewage effluent, litter degradation) | 50-195 MPs·$kg^{-1}$ d.w. |
| [35] | Upper River | Cylinder | Spatio- | River flow | 40 | N/R | 25 L | 10 cm | 4 | 63 µm | Sewage | 6,350 MPs·$kg^{-1}$ |

| | | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | Mersey and Irwell catchments (UK) | resuspension technique | temporal | (flooding), point sources | | | (suspension) | | | | effluent, CSOs | pre-flood 2,812 MPs·kg$^{-1}$ post-flood |
| [21] | River Tame (UK) | Manual (scoop) | Spatial | River flow, point sources, presence of deposition sites (lakes) | 6 | N/R | 250 g | 5-10 cm | 4 | 63 µm | Population density, sewage effluent | 165 MPs·kg$^{-1}$ |
| [29] | River St. Lawrence (Canada) | Ponar grab (0.023 m$^2$) | Spatial | Point sources, GSD, OM | 21 | N/R | 2.4 L | N/R | 3 | 4 µm | Urban land runoff | 65 to 7562 MPs·kg$^{-1}$ d.w. |
| [95] | The North Shore Channel (US) | Ponar grab | Spatial | WWTP | 5 | N/R | 0.75-1 L | 5-10 cm | 4 | 300 µm | Sewage effluent | 36-1613 MPs·L$^{-1}$ |
| [65] | River Kelvin (Scotland, UK) | Manual (spade) | Spatio-temporal | Point sources | 1 | Inner bend | 254.5-441.5 g d.w. | 8-10 cm | 1 | 63 µm | Diffuse sources | 161-432 MPs·kg$^{-1}$ d.w. |
| [46] | River Nakdong (South Korea) | Van Veen grab | Spatial | WWTP | 3 | N/R | N/R | 2 cm | 1 | 20 µm | Sewage effluent | 1970 MPs·kg$^{-1}$ d.w. |
| [161] | River Wei (China) | Grab | Spatial | Point sources, GSD, flow | 15 | Mid-channel and banks | 5 kg | N/R | 3 | 75 µm | Runoff from agricultural lands | 360-1320 MPs·kg$^{-1}$ d.w. |
| [98] | River Elbe (Germany) | Van Veen grab | Spatial | Point sources, flow | 11 | Fine sediment accumulation zones (banks) | 2-4 kg | N/R | 1 | 20 µm | Sewage effluent, industrial emissions | $3.35\times10^6 \pm 6.60\times10^6$ MPs·m$^{-3}$ |

*Where no replicates were mentioned in the study, it was assumed that a single sample per site was collected

N/R – not reported, N/A – not applicable, WWTP – wastewater treatment plant, CSO – combined sewer overflow, GSD – grain size distribution, OM – organic matter, d – depth, MPs – microplastics (count), d.w. – dry weight

Microplastic studies are often criticised for the poor inter-comparison of data between them. Similarly to surface water data, microplastic concentrations reported for sediment samples are difficult to compare, as sampling depths differ between studies depending on the implemented sediment extraction method (Table 2). The top 2-5 cm of sediment is usually captured during manual extraction. Sediment grabs, on the other hand, can sample to as deep as 15 cm depending on the size of instrument used. It is often assumed that most microplastics are present on, or directly below, the riverbed surface [151]. Therefore, analysing deeper sediment layers (which, depending on the depth of the active layer, may contain less microplastics) might have a diluting effect on microplastic concentrations and potentially cause an underestimation of results. Microplastics may be present down to 0.6 m below the riverbed surface, although microplastic depth profiles vary depending on local sediment characteristics [49]. Nevertheless, the majority of studies involve sampling the uppermost layer of the streambed and modifications of sampling protocols may be required to ensure a better data inter-comparison between studies that involve manual sediment collection and the extraction of deeper sediments with grab samplers. For example, Zhang et al. [158] only investigated the top 5 cm of the total sediment collected using a grab. Given the potentially significant discrepancies that can be caused by implementing different sediment extraction techniques, it is necessary that sampling depths be reported in papers.

The amount of sampled material obtained in the studies is rarely reported and varies greatly between studies, from 200 g [157] to 4 kg [25]. For marine studies, NOAA recommends analysing at least 400 g of sediment (wet weight) per sample [176]. While this could be applicable to riverine research, sample size will vary according to sediment characteristics. When sampling coarse-grained substrate, an appropriately larger amount of material must be sampled to obtain a good representation of the fine fraction (<5 mm) that is subsequently analysed for the presence of microplastics. Reporting microplastic concentrations detected in coarse sediments may also require a normalisation of values to a reference mass of the <5 mm fraction, as inter-site variations in microplastic contamination may be concealed by differences in grain size.

Implementing the two contrasting sampling techniques is additionally associated with discrepancies in the units that microplastic concentrations are reported in. Using manual extraction usually involves retrieving material from an area defined by a wooden/metal frame. As a result, concentrations are reported on an area-basis (usually count/mass per $m^{-2}$). In contrast, microplastic abundances are expressed in counts/mass per volume (L, $m^{-3}$) if

material is sampled with a grab. Regardless of the sample collection method used, most authors also express results per kilogram of sediment, allowing for data inter-comparison between studies.

Overall, the role of riverbeds as contamination sinks is currently of major research interest and much progress is being made in the field of sediment sampling for microplastic monitoring. Choosing the suitable sediment collection method will ultimately depend on site characteristics that differ between studies. Nonetheless, care should be taken to report obtained data in units that allow an easy inter-comparison of data, irrespective of sampling protocol implemented.

### 5.4. Considerations for obtaining representative and robust data in microplastic studies

Since protocols for microplastic sample collection and processing have not been unified, varying sampling and analytical quality assurance and control measures have been applied across studies. While ongoing effort is being made to better our understanding of microplastic pollution, best practice should be implemented in every study regardless of the chosen sampling method.

Microplastic contamination is one of the greatest challenges associated with microplastic analysis, and most studies involve taking appropriate precautions to limit the introduction of plastic items into environmental samples during their processing. However, such measures are not normally taken during sample collection. A certain level of airborne contamination will always be associated with lengthy sample handling procedures (e.g. collection of bulk water and subsequent repeated filtration). Some contamination can also be introduced via the sampling equipment utilised and using new equipment is highly advisable due to the tendency of older, weathered plastic items to shed microplastics. Where using new equipment is not possible, the potential contamination can be assessed prior to sampling, e.g. by visual investigation of blank samples under the microscope. Although no specific guidelines exist for on-site contamination monitoring, some authors have accounted for contamination associated with sample collection. For instance, Corcoran et al. [45] collected field blanks parallel to sampling using open Petri dishes containing pre-examined material. Such “blind” samples are usually exposed to air for the duration of sampling (i.e. the time of sample collection and its transfer into a container). It is also good practice to rinse the sampling equipment between subsequent sites [98,177] and stand upstream of the sampling point to

minimise potential contamination from plastic items (e.g. waders) or re-suspended substrate [125].

Similarly, there are currently no clear recommendations specifying what sampled volumes enable a good representativeness of data. As noted by Rios-Mendoza and Balcer, this can lead to an overextrapolation of results and often a sensationalisation of findings [171]. For surface water microplastic studies, sampled areas are typically in the 16-2,840 $m^2$ range. When extrapolated to the widely used $km^{-2}$ (1,000,000 $m^{-2}$) unit, reported concentrations can reach numbers such as 6,800,000 microplastics $km^{-2}$, despite the actual abundances in environmental samples being much lower. For instance, it was calculated that only 3-137 microplastics were found by Mason et al. [119] along the towed distance of approx. 1,400 $m^2$. Once expressed in $km^{-2}$ the reported concentrations reached 2,138-100,016 MPs·$km^{-2}$. Similarly, Lahens et al. [44] reported abundances as high as 172,000-519,000 MPs·$m^{-3}$ (1000 L) based on the recovered 51-140 microplastics per 300 mL of bulk samples. Raw data should therefore be made available where possible [171].

The representativeness of environmental data further relies on taking repeat samples. Taking replicate samples is becoming increasingly common and most studies incorporate collecting an average of 2-3 samples per site. Nevertheless, many studies still fail to mention the quantity of samples or only consider a single sample per site. Collecting repeat samples is especially important when investigating the influence of point sources on microplastic abundance and requires particular attention in wider channels. Mani et al. [26] observed significant variations in microplastic concentrations across a river transect, with much higher pollution levels at the bank where a WWTP outlet was located relative to the mid-channel and the other bank (591,842 MPs·$km^{-2}$ vs 52,364 MPs·$km^{-2}$ and 72,455 MPs·$km^{-2}$ noted for the mid-channel and left bank, respectively). The same was noted for a bank located at a channel confluence with a more polluted stream (201,427 MPs·$km^{-2}$ measured near the confluence relative to 97,498 MPs·$km^{-2}$ detected in another half of the cross-section) [26]. More recently, Wong et al. [160] found that microplastic concentrations in surface waters could vary between the channel and banks by as much as an order of magnitude. A high local variability in microplastic concentrations along a river transect was also mentioned in a study by Dris et al. [112] conducted in the River Seine (France) and occurred irrespective of flow, reinforcing the importance of taking replicate samples during site investigations. The same is noted for studies that investigate microplastic abundance in river sediments. Most authors report taking 2-5 repeat sediment samples per site, which is in agreement with recommendations for marine

research [178]. In rare cases, authors use large quantities of smaller samples; e.g. Klein et al. [25] collected 35-40 replicates from random spots that were later integrated into a single sample. However, it remains rare for authors to describe which morphological unit (riverbank, mid-channel etc.) samples were obtained from. Similar to what is observed in surface waters, microplastic concentrations in sediments vary along the river transect. For instance, Corcoran et al. [45] found significantly higher microplastic concentrations along riverbanks relative to investigated channel centres (441 MPs·kg$^{-1}$ and 276 MPs·kg$^{-1}$ detected in the outer and inner bank, respectively, in comparison to 150 MPs·kg$^{-1}$ found in samples from the mid-channel).

In summary, freshwater microplastic studies are currently of major scientific interest and clear guidelines are urgently needed for microplastic sampling in rivers. Until such recommendations are specified, care should be taken to obtain a good level of representativeness of collected samples in terms of sample volume, location and number of replicates. To facilitate the inter-comparison of studies, enough data should be provided to allow an easy calculation of concentrations in the various units used in microplastic research.

## 6. Conclusion and recommendations for future research

Microplastic studies are in their relative infancy and the environmental transfer of plastic debris between different environmental compartments requires much further investigation. Plastic pollution of riverine systems is of particular interest, as rivers constitute both a major receptor and the principal source of plastic litter to the global ocean. Numerous authors have evaluated the distribution of microplastics in river channels at different scales, and new studies are emerging that describe the temporal changes in microplastic concentrations in rivers. However, most studies focus on the presence of microplastics in surface waters and far less is known about the dynamics of microplastic accumulation and remobilisation from riverbeds, which have been shown to form an essential sink for many particulate microplastics. Sampling techniques are constantly being improved, but the lack of a uniform microplastic definition is reflected in the different methods implemented in the studies, as evidenced by the variable mesh sizes of sampling nets used to capture microplastics. While it is clear that our understanding of the environmental fate of plastic debris is steadily improving, issues remain that require further attention. Therefore, our recommendations for future research are specified below.

**Overall recommendations:**

- Most importantly, a uniform microplastic sampling strategy that specifies their lower size threshold is urgently needed. The International Organisation of Standardisation recently narrowed down the definition of microplastics to 'any solid plastic particle insoluble in water with any dimension between 1 µm and 1,000 µm', while particles >1 mm were classified as large microplastics [179]. However, the current analytical methods do not allow a reliable investigation of particles below 20 µm, and an analysis of such fine fractions is associated with extra costs [180]. Further work is therefore needed to specify an easy to implement size threshold that will enable a harmonisation of sampling procedures, and, in turn, a better inter-comparison between studies [181].
- When deciding on a sampling approach, the main and most important consideration is the cost and benefit of the method [31]. Generally, time- and cost-effective methods are recommended for longer-term monitoring that involves collecting and processing numerous samples, e.g. collecting samples from the riverbank may be chosen over hiring a vessel. However, a standardisation of a cheaper sample collection method would also ensure its greater accessibility and uptake. This is pivotal considering the large emission of plastic waste from rivers flowing through less economically developed countries [182].
- To date, most microplastic studies have been conducted with a relatively low resolution. Presumably, the frequently reported lack of correlation between microplastic concentrations and their point sources in large-scale studies might be due to the large number of variables influencing the transfer of microplastics in riverine systems. Higher resolution investigations may hence yield more reliable results.
- Microplastics partition between the water column and sediments depending on the plastic characteristics, as well as local river morphology and flow regime. However, we are yet to understand what ultimately governs the behaviour and fate of plastic debris. Particularly, the losses of microplastics during their transfer through rivers have not been adequately assessed and the capacity of riverine sediments to act as microplastic reservoirs is largely understudied. Assessing microplastic concentrations in both river water columns and sediments would be highly beneficial for a better evaluation of microplastic transfer in riverine environments and, consequently, their input to the seas and oceans [115]. This is currently made difficult by the different size cut-offs and units used for reporting microplastic concentrations in both matrices (see Table 1 and 2). We therefore recommend that implemented sampling methods should enable a reliable estimation of microplastic abundances in liquid and sediment samples in the same units.

- Authors generally report microplastic concentrations in the unit of count per mass or volume of the investigated medium. However, it has been proposed that microplastic concentrations should be stated using the mass of plastic items, rather than their quantity [183]. Indeed, mass concentrations would be more relevant for the investigation of microplastic mass balance, and may better describe the plastic fragmentation mechanism (as the sum of secondary particle masses will be equal to the mass of the primary item) [59]. On the other hand, smallest microplastics are the most abundant [62]. Weighing plastics under 1 mm in size requires specialised equipment, and individual items may not be detected [183]. Nevertheless, it is advisable to report both mass and count concentrations wherever possible.
- Although a growing body of evidence suggests that microplastic deposition may be analogous to that of natural particles, few authors have so far correlated their findings with riverbed surface grain size distribution [29,52,53]. Understanding the relationship between sediment movement and microplastic behaviour could allow more effective microplastic tracking, and more regular reporting of basic sample characteristics could soon close that gap in knowledge [35,51].
- Greater transparency is needed when describing microplastic sampling techniques. Where possible, it is good practice for authors to provide raw data in the papers and appendices.
- There is a need for reporting much more detailed information about the implemented procedure. Providing information regarding the river morphological units sampled, number of replicates taken, as well as volumes of samples collected, and the quality assurance and control procedures applied will benefit the inter-comparison of studies.

**Specific recommendations:**

- The most effective way of mitigating microplastic pollution is to limit their at-source emission. While assessing their input from diffuse sources remains challenging, it is much easier to establish the contribution from point sources. Several studies have already described the emission of microplastics from WWTPs [29,97,159]. However, other point sources have not been evaluated in equivalent detail (e.g. plastic manufacturing plants). More detailed sampling (upstream and downstream, accounting for intra-site variations) combined with a thorough analysis of microplastic characteristics (e.g. morphology/extent of weathering, polymer type) and associated contaminants (organic compounds, metals and biofilms) can facilitate tracking of the various sources of microplastics to river systems [184].

- Although it is well described that microplastics accumulate in zones of impeded flow, the role of morphological barriers such as mid-channel bars in microplastic retention has not been explored in much detail [41]. Presumably, plastic debris could be entrained and buried into bedforms such as dunes and later remobilised during their passage [185], adding further complexity to the interplay between microplastic and sediment dynamics. The influence of river planforms (e.g. meandering or braided) on microplastic distribution has also been neglected. Braided rivers are characterised by a high stream slope and substantial variations in water discharge. Therefore, their temporal microplastic inputs into adjacent seas may exceed those noted for meandering rivers. This is especially important considering that some of the most polluted rivers, such as the Brahmaputra (India), are braided and contain predominantly fine sediment that can be easily mobilised [94]. Given the presumably large impact of riverbed formations and planforms on microplastic flux, river morphology should receive more attention and its role should be accounted for in future models of microplastic emissions to the global ocean.
- Most studies to date described the spatial distribution of microplastics in riverine environments. Although more studies are emerging that assess the spatio-temporal trends in microplastic abundance in both river surface waters and sediments, much more effort is needed to understand the impact of different flow rates on microplastic flux into the adjacent seas and oceans. When investigating microplastic concentrations on a temporal scale, data should be correlated with variables such as river flow/discharge, precipitation or water levels. Such data can often be accessed online free of charge and, given the large influence of flows on microplastic transfer, should be utilised in future research.
- Due to the multitude of variables affecting both natural and plastic particles in river channels, studying the fate of microplastics in natural settings is challenging and supporting field data using an experimental approach may be beneficial. Using hydrological flumes, or mesocosms enables an investigation of microplastic response to a single variable under controlled conditions and may provide answers to the most pressing research questions. On the other hand, because of scale limitations and the frequently noted lack of repeatability and reproducibility associated with experimental studies, continued effort is needed to understand the environmental fate of microplastics through field monitoring.

## Declaration of Competing Interest

The authors declare no conflict of interest.

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