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# A source-to-sink perspective of an anthropogenic marker: A first assessment of microplastics concentration, pathways, and accumulation across the environment

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## ABSTRACT

Source-to-sink geoscientific domain and environmental plastic cycle studies are two major scientific worlds starting to interact, taking benefit from each other. To advance in our understanding of the sharing benefits between interconnecting research communities, we firstly carry out a review from sedimentology, sources, sinks, transport dynamic and pathways of microplastics along the entire source-to-sink (S2S) profile. The main peculiarities for microplastics are the numerous and distributed sources across the environment, as well as the importance of physical properties and shape factors. Then, we propose a review of plastic mass concentration along the S2S profile to discuss influence of sedimentation rate on microplastic accumulation and to identify intermediate reservoir and final fates. Deep sea deposits, including turbidite systems are potentially hotspots for microplastic accumulation that are poorly studied, deserving much more attention to scale mass balance studies. This review finally highlights areas of synergies between S2S geoscientific and plastic communities to guide future interdisciplinary microplastic research. Most of these issues will rely on multiplying measurements across all matrices and environments based on standard technology to generate homogenized measurements and large database for plastic and microplastics.

## 1. Introduction

Particulate plastic pollution is ubiquitous, with plastics found along with natural sediments within all depositional systems, from soils, fluvial sediments, beaches, deltas, to the deep sea.

Source-to-sink (or S2S) studies are built on the concepts of the sediment routing systems (e.g., Galloway et al., 2000; Martinen et al., 2010; Allen, 2017), and aims at understanding the spatial and temporal dynamics of sediments within a global or closed system by quantifying fluxes and budgets. The transport of a grain of sand through the sediment routing system is complex, with potential changes in the size, and shape of the grain, temporary storage within the fluvial system until eventual permanent geological storage in the final sink (e.g., Allen, 2008). To understand what is written within the layers of sediment deposited within a sedimentary basin, or sink, it is important to understand the routing system that got the sediment from its source and deposited in the sink, notably using sedimentary budgets (e.g., Hinderer, 2012; Rohais et al., 2021). This encompasses understanding the source

locations, the transport dynamics of the sedimentary particles within the fluvial systems en-route from that source, and how the particles might be altered or temporarily stored en-route. This leads to the buffering, alteration, or amplification of signals of change in the source region. Furthermore, regions of temporary storage might act like capacitors, building up a stock of grains that will cascade into the sink when a certain event tips the balance. Concepts such as buffering, temporary storage and bypass are key components to S2S, and can be used to understand where sediments will accumulate.

The transport of plastic in the environment seems even more complex. First plastic particles span wide ranges of density and shapes. Second, plastics will fragment and degrade into microplastic particles that will be transported through the same sediment routing system from their source to eventual sink. The transport of microplastics has similarities with aspects of different sediment grains, yet likewise strong differences (e.g., Waldschläger et al., 2022). Like sediments, accumulations of microplastics will be subject to buffering, temporary storage, and by-pass as they cascade along the routing system. Our

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at least with such four classes.

## 2.2. Shape and type

The shape categories used for both microplastics and mesoplastics include pellet, fragment, fiber, film, foam, and other (Fig. 1) (e.g., Crawford and Quinn, 2017; Burns and Boxall, 2018). These categories are beneficial for environmental monitoring and for tracing the sources of plastic particles (Rochman et al., 2019; Rosal, 2021). However, for small microplastics and nanoplastics, the shape categories are often limited to fiber and fragments due to the fragmentation processes and the constraints of associated analytical imaging techniques. Waldschläger et al. (2022) proposed that these shape categories should be reevaluated, especially considering that particle shapes within certain categories, such as pellets, can vary significantly, ranging from cylindrical to lenticular or disc shaped.

Shape is effectively a fundamental characteristic that influences particle dynamic from entrainment, transport, to deposition (e.g., Dietrich, 1982; Allen, 1985). Shape metrics in natural sediments include for example long axis length (L), intermediate axis (I), short axis (S), perimeters, and rugosity (e.g., Chamley, 1990). They are used to quantify shape descriptors such as the Corey Shape Factor (CSF, Corey, 1949) ( $SL^{-1/2}T^{1/2}$ ) often referred to for roundness of particle (Dietrich, 1982). van Melkebeke et al. (2020) suggested taking advantage of such routine sedimentological descriptors to tackle challenges in microplastic studies. While it seems feasible for LMP, analytical challenges in imaging small particles should be overcome to gain insights on these issues (e.g., Cowger et al., 2020).

In 1988, the Society of the Plastics Industry introduced the Resin Identification Code (RIC) system which divided plastic resins into seven different categories, or types (Fig. 1). They correspond to the most common plastics distributed across the environment (Andrady and Neal, 2009). Plastics contain organic polymers primarily formed from chains of carbon and hydrogen atoms (Fig. 1), influencing their capacity to degrade across the environment as well as to be transported (density).

## 2.3. Density

The density of plastic particles range in the values that are similar to

organic matter, the upper range of the modern plant substances and woods, and the lower range of sedimentary organic matter (Fig. 2). They are however much lower than for rocks, minerals, and sediments, characterized by a mean value assumed to be  $2.65 \text{ g/cm}^3$  as for quartz (Fig. 2) (Chamley, 1990; Tyson, 1995; Sharma, 1997). The density of microplastics varies between  $<0.05 \text{ g/cm}^3$  for expanded polystyrene and  $2.3 \text{ g/cm}^3$  for polytetrafluoroethylene (Teflon) (Chubarenko et al., 2016 and references herein; website OMNEXUS). Of a sample of 188 different plastic polymers, the range of densities has a minimum 1st quartile of  $1.1 \text{ g/cm}^3$  and a maximum 3rd quartile of  $1.46 \text{ g/cm}^3$  (Fig. 2). Most plastics therefore have a density greater than that of water, whether it is sea water or freshwater. Many common types of polymers, such as PET (polyethylene terephthalate), PVC (polyvinyl chloride), and PS (polystyrene), inherently have a higher density than water. Additionally, the density of plastics can be further increased by the addition of additives, such as fiberglass, which are used to enhance the properties of the plastic. There are relatively few types of plastics that have lower densities than water. Two notable examples are PE (polyethylene) and PP (polypropylene). Interestingly, these two polymers, which have lower densities, are among the most dominant types of plastics found in the natural environment resulting from mismanaged plastic waste, especially in the marine domain (e.g., Erni-Cassola et al., 2019).

An interesting attribute of microplastics is that their density will change as the particle is transported within the water column. This will mean that light particles may be transported along great distance in suspension to subsequently settle as their structure is altered due to weathering, degradation, and the colonization of the particles. The colonization of microorganisms associated with other organic particles attaching to the surface of microplastics (biofouling) can either increase or decrease the density of microplastic particles, especially depending on the nature of the associated organic particle (Rummel et al., 2017). Fragmentation and degradation of microplastics could also result in a decreasing density by creating additional pore volume inside the particles (Ter Halle et al., 2016). However, currently such density changes are still poorly quantified (see Skalska et al., 2020).

## 3. Physical properties of transport of microplastics

Although some microplastics may be airborne transported as dust

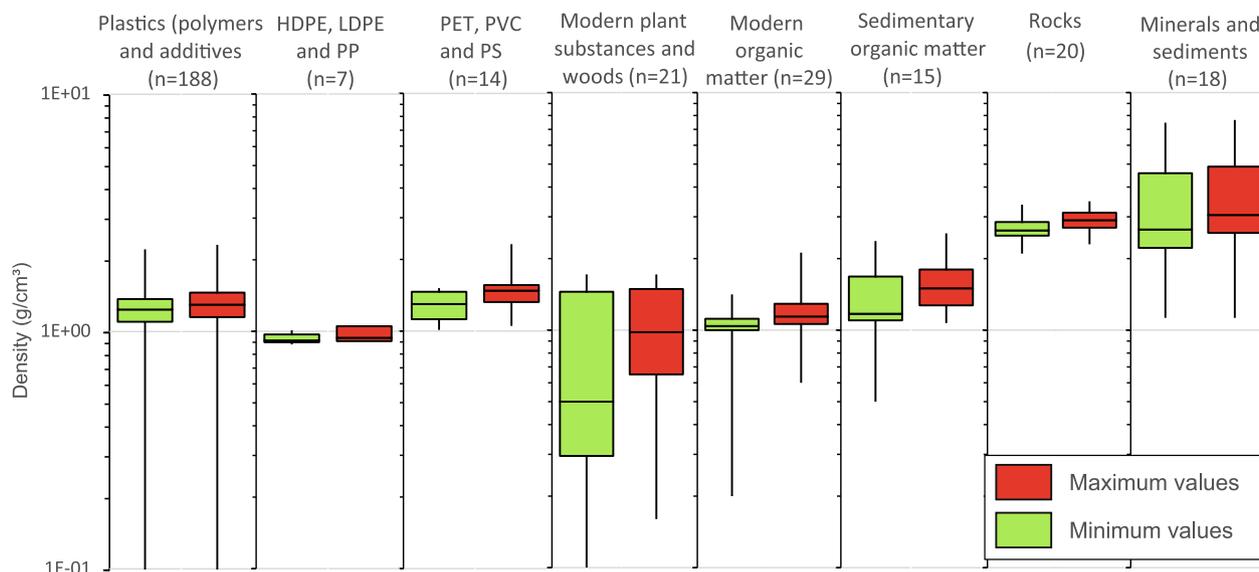


Fig. 2. Synthesis of 188 values of plastic density ( $\text{g/cm}^3$ , log scale) presented in box plot (1st to 3rd quartile) and compared with other natural particles including modern plant substances and wood, modern organic matter, sedimentary organic matter, rocks and minerals and sediments. Green box plots illustrate the minimum values, and red box plots the maximum values. Data for plastic are from Omnexus website (<https://omnexus.specialchem.com/polymer-properties/properties/density>) and Chubarenko et al. (2016) and references herein. Data for organic materials are from Tyson (1995) and references herein. And data for rock and minerals are from Sharma (1997) and references herein. (For interpretation of the references to color in this figure legend, the reader is referred to the web version of this article.)

particles, the main transport path for microplastics is considered to be rivers, as is the case for natural sediments (Hay, 1998). Keeping this in mind this review focuses on the main physical processes impacting the transport of microplastics along the fluvial system. At this point, it is important to mention that the present review is solely dedicated to microplastics and their transport behavior. In contrast to microplastics, the transport behavior of nanoplastics, (particles with a size at the nanometric scale,  $< 1 \mu\text{m}$ , Fig. 1) is mainly characterized by Brownian motion in the fluid phase, that dominates over buoyancy and sedimentation (Gigault et al., 2021). Consequently, nanoplastic particles detected in sediments mostly originate from fragmentation of larger plastic debris along the producer and consumer chains.

The transport of microplastics within the sediment routing system is primarily a function of how these particles are transported by flowing water and is dependent on various physical properties such as the grains size, density, and the thresholds for motion. While microplastics are man-made polymers, they are also solid particles like any other sediment, and arguably fall within the field of physical sedimentology according to the definition of Allen (1985). In this perspective, several recent studies have illustrated the similarities between plastic and sediment grains, whether on descriptive aspects (grain size, shape, type, density), or on transport processes (e.g., Kane and Clare, 2019; Harris, 2020; Waldschläger et al., 2022).

Transport of plastic particles within the water column can reduce to two physical processes: the impact of the stress of the flow of water acting on the particle that drives the particle into motion, “mobilization”, and the subsequent settling of the particle within the water column, “buoyancy”. Plastic particles will, in simple terms, either travel along the bed, as bedload, or become suspended in the water column and travel as suspended load. We therefore break the description of transport dynamics into two parts, mobilization and buoyancy.

### 3.1. Microplastic mobilization

The Shields’ diagram (Fig. 3) is a valuable tool to estimate the threshold of motion (Shields, 1936). The Shields’ diagram is based on dimensional analysis and fluid mechanics considerations and represents the relation between the ratio of the bed shear stress and the gravitational force on a particle (Shields parameter,  $\theta = \frac{\tau}{g(\rho_s - \rho)d}$ ) as a function

of the boundary Reynolds number ( $Re^* = \frac{u_*^* d}{\nu}$ ).  $\tau$  is the bed shear-stress at the threshold of motion,  $g$  is acceleration due to gravity,  $\rho_s$  is density of the sediment,  $\rho$  and  $\nu$  are the density and kinematic viscosity of the water,  $d$  is the sediment grain diameter, and  $u_*^*$  is the threshold friction velocity.

Wright and Parker (2004) completed the Shield’s diagram to include the limit for significant suspension between suspended load and bedload (Fig. 3). Several studies have shown a relationship between microplastic abundance and sediment grain size (Vianello et al., 2013; Ballent et al., 2016; Maes et al., 2018; Enders et al., 2019). Meanwhile, additional studies have also measured resuspension thresholds for microplastics, revealing that particle shape, size, and the size distribution of sediment play a significant role in their resuspension behavior (Ballent et al., 2012; Waldschläger and Schüttrumpf, 2019a).

In the Shields’ diagram (Fig. 3), microplastics show a larger variability than natural sediments. Some points with a smaller shields number than natural sediments indicate that microplastics may have a higher mobility compared to similar-sized sediment grains (Fig. 3). Other points seem to indicate the opposite in the case of pellets of fragments (Fig. 3). It seems that adding natural sediment to microplastic helps their onset of motion as illustrated by the positive trend in the shield diagram (Fig. 3) (Waldschläger and Schüttrumpf, 2019a). Data on microplastic erosion dynamics is nevertheless limited compared to sediment dynamics research from the past (e.g., van Rijn, 1984a).

To develop a more comprehensive understanding of microplastic resuspension behavior, further in-depth studies are required. These studies should encompass a wider range of microplastic parameters and explore different definitions of the onset of motion in order to derive general expressions for microplastic resuspension behavior.

### 3.2. Microplastic buoyancy

The second main parameter to constrain transport dynamic of microplastics is their buoyancy. The vertical concentration of suspended microplastics can be predicted using the Rouse approach (Cowger et al., 2021). This theoretical model is based on the settling and rising velocities. Various studies have been conducted to determine these velocities for different microplastics through physical experiments (Fig. 4) (van Melkebeke et al., 2020; Khatmullina and Isachenko, 2017; Kaiser et al.,

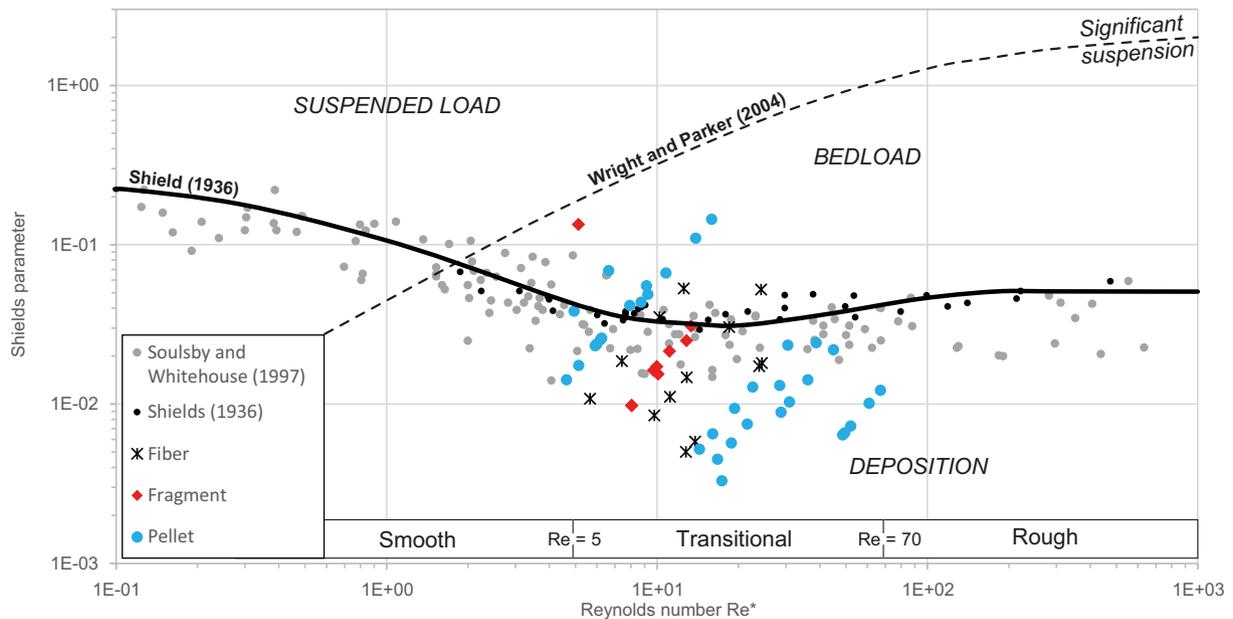
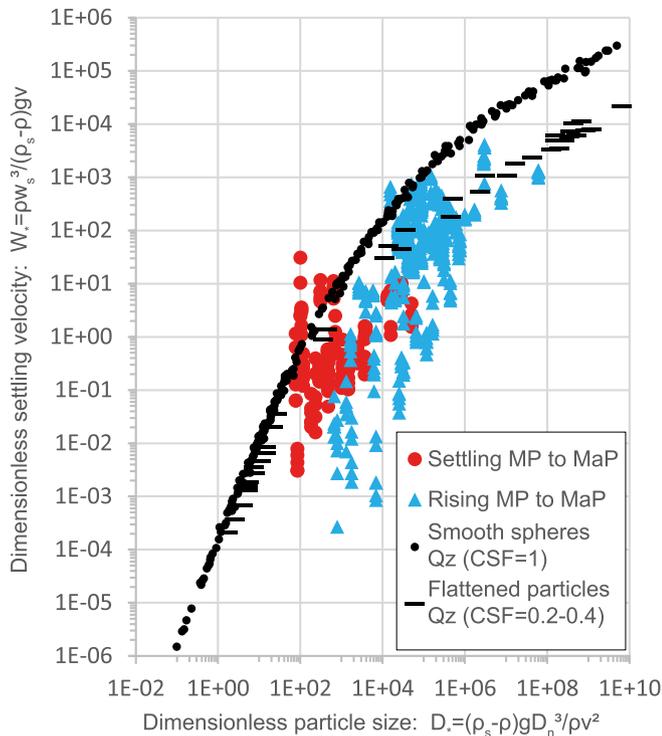


Fig. 3. Shield diagram with the original dataset (black dots) (Shields, 1936) and dataset by Souldsby and Whitehouse (1997) for fine-grained and potentially cohesive sediment (gray dots). Microplastic data are plotted and computed from the experiments form Waldschläger and Schüttrumpf (2019a). It seems that microplastics have higher mobility compared to similar-sized sediment grains.



**Fig. 4.** Dimensionless settling velocity versus dimensionless particle size of microplastics (MP) to macroplastics (MaP) (Red dots and blue triangles) compared to the reference dataset of [Dietrich \(1982\)](#) for smooth spheres and flattened quartz (Qz) particles (black dots and rectangle, respectively), where CSF corresponds to the Corey shape factor ([Dietrich, 1982](#)). Microplastic data are computed from the experimental results of [Waldschläger and Schüttrumpf \(2019b\)](#). For an identical dimensionless particle size, the settling velocity of microplastic is primarily smaller than flattened quartz particle, itself being smaller than spherical quartz grains. (For interpretation of the references to color in this figure legend, the reader is referred to the web version of this article.)

[2019](#); [Waldschläger and Schüttrumpf, 2019b](#); [Waldschläger et al., 2020](#)). [Waldschläger and Schüttrumpf \(2019b\)](#) showed that theoretical approaches used in classical sediment transport for calculating settling velocities are inadequate for microplastics. Instead, shape-dependent drag models appear to provide a better fit ([van Melkebeke et al., 2020](#)). This suggests that the shape of microplastics may play a more significant role in their transport compared to natural sediments ([Fig. 4](#)).

The explanation for this property becomes apparent through an order of magnitudes analysis. For a small sphere settling in a quiescent fluid, the drag force is proportional to  $F_d \sim \mu UD$  where  $\mu$  is the fluid viscosity,  $D$  is the sphere diameter and  $U$  the steady state settling velocity ([Guazzelli and Morris, 2011](#)). Balancing this drag force with the buoyancy force, which scales as  $F_b \sim (\rho_p - \rho)gD^3$  where  $\rho_p$  is the particle density,  $\rho$  is the fluid density, and  $g$  is the acceleration of gravity, we obtain the classical scales for sedimentation velocity  $U \sim (\rho_p - \rho)gD^2/\mu$ .

The situation is different for elongated fibers. In this case the drag force scales as  $F_d \sim \frac{\mu UL}{\ln(\frac{L}{b})}$  where  $L$  is the length of the fiber ([Batchelor,](#)

[1970](#)). Consequently, the leading-order expression for the settling velocity of the fiber becomes  $U \sim \frac{(\rho_p - \rho)gD^2}{\mu} \ln\left(\frac{L}{b}\right)$ . Therefore, the correct scaling for the sedimentation velocity for a fiber is based on its diameter. Making use of the fiber's equivalent diameter based on a sphere of the same volume may lead to significant errors in velocity predictions. This notion is further supported by the higher variation of aspect ratios (longest/shortest side length) observed in microplastics compared to

natural sediments ([Kooi et al., 2018](#); [van Hateren et al., 2020](#)).

Deformable microplastic particles such as foils (e.g., films) behave differently during settling than rigid ones due to their ability to change shape during transport ([Waldschläger et al., 2020](#)). In particular, flexible fibers may deform significantly during sedimentation, modifying their velocity ([Marchetti et al., 2018](#)). However, more research is needed to fully understand and compare the deformability of different types of microplastics taking benefit of knowledge from natural particles ([Ardekani et al., 2017](#)).

### 3.3. Aggregation, flocculation, and biofouling

Both sediments and microplastics may experience changes in settling behavior due to flocculation and aggregation, resulting in enhanced deposition of fine particles in aquatic environments ([Williams et al., 2008](#); [Alimi et al., 2018](#); [Wang et al., 2021a](#)). However, despite its significance, the understanding of flocculation and its effects on microplastics is still not well-explored.

Coarse mineral particles (e.g., sand, gravel) are typically transported individually, without significant aggregation or cohesion. Fine, cohesive sediments (e.g., silt to mud) are however typically transported in association with other mineral and organic materials as aggregates or flocs ([Droppo et al., 1997](#)). Floc formation is influenced by several factors, including the frequency and energy of particle interactions driven by concentration and hydrodynamic conditions, the electrostatic charge of particle surfaces, and the concentration and character of dissolved ions. Floc formation tends to increase with higher salinity ([Winterwerp et al., 2006](#); [Winterwerp, 1998](#)). The resulting flocs are larger than the size of their constituent mineral grains and generally have lower densities, ranging from 1.0 to 1.4 g/cm<sup>3</sup> ([Droppo, 2001](#); [Faure et al., 2015](#); [Maggi, 2005](#)).

When microplastics aggregate with suspended sediments and metal oxides, it results in higher aggregate density ([Wu et al., 2019](#); [Leiser et al., 2020](#)). However, marine aggregation with other substances causes a decrease in aggregate density compared to individual microplastic particles. While higher salinity is believed to increase microplastic flocculation, the influence of microplastic particle properties on floc formation remains unstudied. Microplastic floc formation is described using collision frequency and attachment efficiency based on natural sediment floc formation and preliminary physical experiments ([Besseling et al., 2017](#); [Del Domercq et al., 2021](#)).

Biofouling is a successive buildup of organic matter and organisms (e.g., [Katija et al., 2017](#)). It also plays a crucial role in altering the buoyancy and settling behavior of plastic particles ([Zettler et al., 2013](#); [Kooi et al., 2017](#)). It can lead to increased sinking rates of plastic particles ([Kaiser et al., 2017](#)), especially those with large surface area to volume ratios ([Ryan, 2015](#)), influencing their fate in the marine environment ([Fazey and Ryan, 2016](#)).

## 4. Sources and sinks of microplastics

In terms of sources and sinks, microplastics have some commonalities with sediment grains of equivalent size: a continental source, transported by water downstream and eventually down to the deep marine system. Only a small proportion is transported in the air and distributed over the landscape by rainfall.

### 4.1. Sources

Unlike natural sediments derived from erosion in the catchment, or from previous deposits, plastics have an anthropogenic origin and enter the natural environment as either distributed particles across the landscape or from point sources directly into the fluvial system ([Table 1](#) and [Fig. 5](#)). Thus, sources of microplastics are distributed within river catchments depending on human activities and waste management systems. In this review, "source" is meant as the location where the

**Table 1**

List of some primary and secondary sources of microplastic that are released into the natural environment. See Fig. 5 for a schematic representation.

Nb. in Fig. 5	Primary or secondary microplastic	Description and examples	Type of source within the sediment routing system
1	Secondary	Release of microplastics from managed waste treatment and storm run-off. Examples are plastic fibers and plastics in cosmetics where they do not settle out (see numbers 6 and 8).	Point source
2	Primary	Nurdles, which are the base plastic pellets used in manufacture of solid plastic objects. This should only enter the natural environment through spills.	Point source
3	Secondary	Microplastic through activities related to both blasting and due to exterior paints, that contain plastic (Takahashi et al., 2012; Song et al., 2015; Chae et al., 2015).	Point source
4	Secondary	Microplastic from fly tipping and unmanaged or poorly managed waste disposal.	Point source
5	Secondary	Transport causes the release of particle of rubber that will enter the environment both directly into the air and as run-off into the rivers (Kole et al., 2017; Evangelidou et al., 2020).	Distributed source
6	Primary	Microbeads used in hygiene and cosmetic products. In some countries this plastic is treated and removed from waste waters (Murphy et al., 2016).	Point source
7	Secondary	Plastic waste due to human activity and the use of landscape by the tourism industry that could cause a locus of littering and release of synthetic fibers and plastics from abrasion.	Distributed source
8	Secondary	Fibers from textiles, usually released into the environment through wear and tear and washing (Habib et al., 1998; Browne et al., 2011; Napper and Thompson, 2016; Boucher and Priot, 2017).	Point source
9	Secondary	Agricultural use of plastic and local tipping of plastic is a distributed source of microplastic across farmed land. A common example is fragments of plastic that cover silage bales.	Distributed source
10	Secondary	Release from landfill, all types of plastic.	Point source
11	Secondary	Fishing can lead to the dumping of plastic through the loss of synthetic fishing nets and lines that break down into microplastics (Lusher et al., 2017). This plastic waste bypasses the terrestrial routing system.	Distributed source
12	Secondary	Shipping waste despite the dumping at sea being internationally band. This	Distributed source

**Table 1 (continued)**

Nb. in Fig. 5	Primary or secondary microplastic	Description and examples	Type of source within the sediment routing system
13	Secondary	plastic waste bypasses the terrestrial routing system. Waste from marine platforms, which is banned internationally. This plastic waste bypasses the terrestrial routing system.	Point source

plastics enter the natural routing system, not the point at which the plastic leaves the cycle from primary plastic to waste. For GESAMP (2016), source of plastics generally groups into four categories: producers and converters, sectoral consumers, individual consumers, and waste management.

In 2015 it was estimated that 407 Mt of primary plastic (virgin plastic created from raw materials) entered into a use phase, while 302 Mt left the use-phase (Geyer et al., 2017). Plastic that is not contained within managed landfill is termed mismanaged plastic waste (MPW) and estimated between 65 and 99 Mt in 2015 (Lebreton and Andrady, 2019). Of this, between 0.8 and 4 Mt are estimated enter the ocean each year from rivers (Lebreton et al., 2017; Schmidt et al., 2017; Meijer et al., 2021).

The estimates of plastic waste that enters the ocean differ on the assumptions for the spatial distribution of MPW and the impact of climate on river run-off on the delivery of plastic to the oceans. The location of release is varied, with some such as the particles of tire rubber being semi-distributed into both the atmosphere and as run-off from roads directly into the fluvial network, while others such as fibers from synthetic materials being released more directly into the fluvial network. Table 1 and Fig. 5 outline some of these sources both in terms of their manufacturing and use cycle and the processes that would lead to their release as a source in the source-to-sink framework.

Sampling of rivers globally for microplastics is however patchy, and there is large variation in the quantities measured (e.g., Rodrigues et al., 2018; Eo et al., 2019; Constant et al., 2020). Consequently, the importance of small river catchments versus large rivers is uncertain. This uncertainty is demonstrated by for example a local study of microplastic contamination within the Mersey and Irwel rivers in England. These two small rivers were found to contain roughly half a million particles of microplastic per square meter, which would make them one of the most polluted rivers globally (Hurley et al., 2018). This watershed however does not suffer from open tipping of plastic, and there is no strong evidence for mismanaged water treatment. This raises two questions: (1) is there a sampling bias and microplastic concentrations might be very high next to the point sources, where treated and untreated waste enters the fluvial system, or (2) is there significant temporary storage within fluvial systems (see for example van Emmerik et al., 2022).

The quantity of microplastics entering the rivers can be significantly reduced through wastewater treatment, however microplastics related for fabrics are still transported into the rivers (Woodward et al., 2021). From looking at samples from riverbank deposits, it was found that in the River Tame, UK, microplastic is sourced from both treated wastewater and more importantly untreated waste. The evidence from this catchment would suggest that there are point sources of continuous microplastic from urban treatment works and factories. Flood waters will however source microplastics that have become deposited within the river catchment and hence there are events where microplastic is sourced catchment wide. In a similar study on the Brisbane River, Australia, it was found that the concentration of microplastics deposited at the riverbank did not vary spatially along with land use, however the types of plastics found vary (He et al., 2020). In rural areas microplastic deposition was dominated by polyethylene (He et al., 2020).

In water treatment stations, the dense microplastics are typically

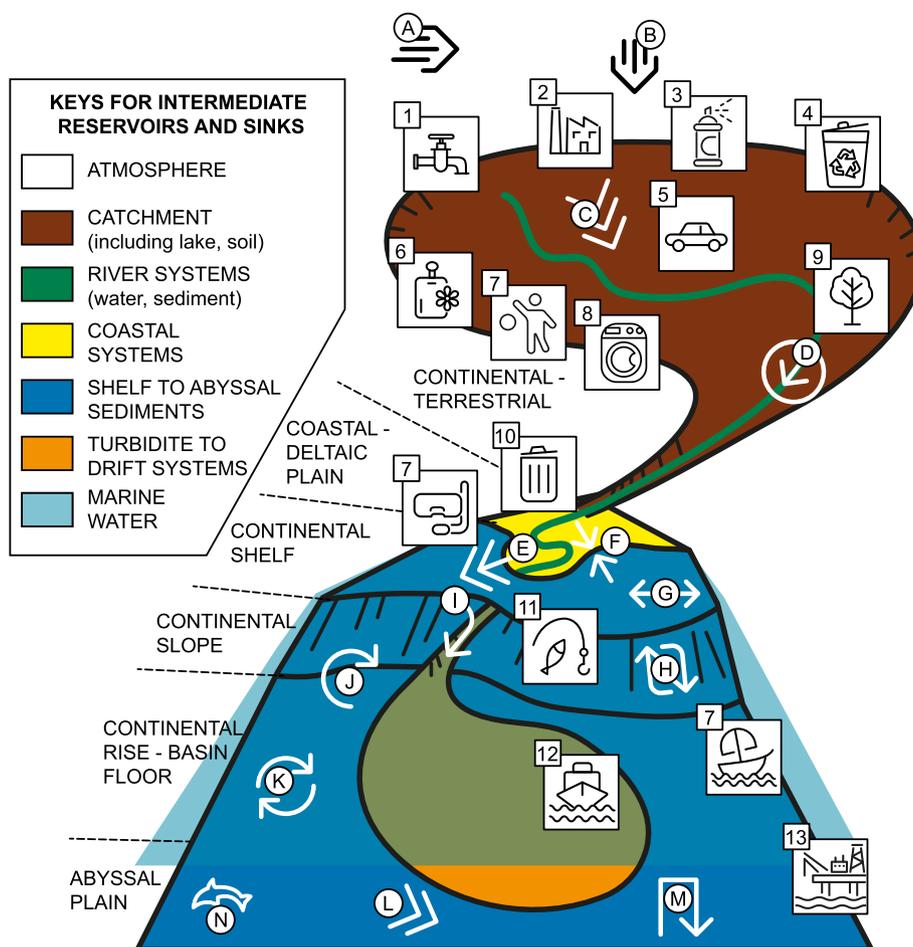
**Table 2**  
Listing of datasets presented in Figs. 9 and 10.

Compartment	Matrice	Environment	Consulted references (Nb.)	Selected references (Nb.)	Reference list	Nb. of measurements	Concentration (ppm)					
							Min.	1st Quartile	2nd Quartile	3rd Quartile	Max.	Mean
Atmosphere	Air		73	5	Liu et al. (2019)	6	1.42E-09	8.95E-09	3.00E-07	1.37E-06	3.40E-06	8.04E-07
					Fauser et al. (1999)	1						
					Schauer et al. (2002)	1						
					Amato et al. (2014)	1						
					Fomba et al. (2018)	4						
					<b>Total:</b>	13						
					Liu et al. (2019)	67						
					Kumata et al. (2000)	4						
					Kumata et al. (2002)	4						
					Zakaria et al. (2002)	2						
					<b>Total:</b>	77	6.33E-02	1.49E+01	4.81E+02	3.78E+03	2.04E+05	5.24E+03
Lithosphere	Dust	Catchment	73	4	Corradini et al. (2019)	15						
			33	15	Bläsing and Amelung (2018)	2						
					Dierkes et al. (2019)	5						
					Gajst (2016)	1						
					Fuller and Gautam (2016)	3						
					Ljung et al. (2018)	3						
					Rezaei et al. (2019)	10						
					Vollertsen and Hansen (2017)	6						
					Zhang et al. (2018)	3						
					Zhang et al. (2020a, 2020b)	10						
					Li et al. (2020)	4						
					Braun (2022)	16						
					Müller et al. (2022)	22						
					Kocher et al. (2008)	4						
					Panko et al. (2013)	3						
					<b>Total:</b>	107	1.00E-05	1.50E+00	4.40E+01	6.74E+02	1.58E+05	3.77E+03
	Soil Sediment	Lake	16	1	Faure et al. (2015)	43						
								<b>Total:</b>	43	1.25E-02	6.75E-01	5.63E+00
		River	34	9	Eo et al. (2019)	1						
							Enamul Kabir et al. (2022)	28				
					Fan et al. (2019)	6						
					Rodrigues et al. (2018)	6						
					Rao et al. (2020)	7						
					Niu et al. (2021)	5						
					He et al. (2020)	53						
					Klein et al. (2015)	12						
					Sarkar et al. (2019)	9						
					<b>Total:</b>	127	6.80E-04	3.50E+00	1.32E+01	4.60E+01	9.34E+02	5.33E+01
		Beach	34	8	Martins and Sobral (2011)	6						
							Tiwari et al. (2019)	3				
					Claessens et al. (2011)	8						
					Cadiou et al. (2020)	8						
					Jayasiri et al. (2013)	72						
					González-Hernández et al. (2020)	50						
					Aslam et al. (2020)	16						
					Fok et al. (2017)	8						
					<b>Total:</b>	171	5.00E-02	5.68E+01	1.50E+02	2.80E+02	2.42E+03	2.37E+02
		Coastal to Shelf	19	2	Reddy et al. (2006)	12						
							Claessens et al. (2011)	17	8.40E-01	1.22E+00	3.34E+00	8.07E+01

(continued on next page)

Table 2 (continued)

Compartment	Matrice	Environment	Consulted references (Nb.)	Selected references (Nb.)	Reference list	Nb. of measurements	Concentration (ppm)						
							Min.	1st Quartile	2nd Quartile	3rd Quartile	Max.	Mean	
Hydrosphere	Water	Slope	8	1	Total:	29							
					<a href="#">Jones et al. (2022)</a>	6	1.00E-03	7.75E-03	1.00E+01	6.83E+02	1.95E+03	3.72E+02	
		Canyon	8	1	Total:	6							
					<a href="#">Jones et al. (2022)</a>	8							
		Lake	16	1	Total:	8							
					<a href="#">Faure et al. (2015)</a>	29	7.40E+02	1.09E+03	2.91E+03	3.50E+04	1.54E+05	2.65E+04	
		River	34	11	Total:	29	1.56E-02	2.50E-02	5.56E-02	2.56E-01	4.28E-01	1.45E-01	
					<a href="#">Eo et al. (2019)</a>	24							
					<a href="#">Fan et al. (2019)</a>	12							
					<a href="#">Haberstroh et al. (2021)</a>	80							
<a href="#">Kataoka et al. (2019)</a>	36												
<a href="#">Treilles et al. (2022)</a>	12												
<a href="#">Rodrigues et al. (2018)</a>	9												
<a href="#">Hohenblum et al. (2015)</a>	65												
<a href="#">van der Wal et al. (2015)</a>	6												
<a href="#">Moore et al. (2011)</a>	9												
<a href="#">Faure et al. (2015)</a>	14												
<a href="#">Lechner et al. (2014)</a>	2												
Total:	269	1.00E-09	8.20E-05	6.70E-04	1.47E-02	4.04E+01	2.69E-01						
Shallow marine	19	1	Total:	24									
			<a href="#">Sonke et al. (2022)</a> (Ref. herein)	24	3.71E-08	3.12E-06	2.78E-04	9.65E-04	3.23E-03	5.89E-04			
Marine			Total:	24									
			<a href="#">Sonke et al. (2022)</a> (Ref. herein)	36									
			<a href="#">Pabortsava and Lampitt (2020)</a>	36									
			<a href="#">Goldstein et al. (2012)</a>	836									
			<a href="#">Doyle et al. (2011)</a>	595									
Total:	1503	9.80E-10	9.80E-09	1.96E-07	3.77E-04	2.45E-01	2.24E-03						
Deep marine	9	1	Total:	10									
			<a href="#">Sonke et al. (2022)</a> (Ref. herein)	10	1.80E-07	3.77E-05	2.16E-04	6.89E-04	9.89E-04	3.62E-04			
Total:	10	1.80E-07	3.77E-05	2.16E-04	6.89E-04	9.89E-04	3.62E-04						



**Fig. 5.** 3D schematic model of the plastic cycle integrated in the sediment routing systems and related transport pathways (arrows and labelled with letters) across the main intermediate reservoirs and the final deep marine sink. 14 transport pathways are represented (A to N). Sources of plastics are represented using icons in white squares and labelled with numbers. 13 plastic sources are represented with: 1- water and wastewater treatment plants, 2- manufacturing and industries, 3- construction and painting, 4- solid waste management, 5- road and transport, 6- cosmetics and personal care products, 7- tourism and recreational activities, 8- textile and clothing, 9- agricultural activities, 10- littering and landfills, 11- fishing activities, 12- shipping, 13- offshore industries. See [Table 1](#) for further information.

removed through settling and remain in the sludge. Microplastics within fabrics are significantly harder to remove, hence they seep into the rivers along with the treated water. The sludge is however processed and used in many countries as a fertiliser. This raises the potential that microplastic enters rivers from runoff withing agriculture land (Nizzetto et al., 2016), and might explain the lack of variability in microplastic concentrations in the Brisbane River.

Estimates of total plastic show an exponential growth in production. Projections of future quantities of microplastic entering the ocean via rivers would suggest that most of the plastic will come from Asia and Africa (e.g., Lebreton et al., 2017; Nyberg et al., 2023). This projection is based on inferred relationships between the mismanagement of all waste and the GDP of countries mapped on to population grids (e.g., Lebreton and Andrady, 2019). Subsequently empirical model is used to relate the flux of microplastics to the sediment flux. This correlation remains poorly tested (e.g., Lebreton et al., 2017; Weiss et al., 2021) as there is no current large database of the quantities of plastic that are released from all rivers.

Unfortunately, with the current lack of continuous monitoring within any fluvial system current estimates on the sources of microplastics are extrapolated from only a handful of measurements.

#### 4.2. Intermediate reservoir and sinks

Plastic sinks are not primarily controlled by human activity, as the

microplastics accumulate in the same zones where natural sediments accumulate, particularly on beaches, deltas and deep marine seafloor.

The plastic sinks encompass four primary domains: hydrosphere, lithosphere, atmosphere, and biosphere, which are commonly referred to as matrices of water, sediment, air, and biota. Atmosphere could be primarily classified as intermediate reservoir as microplastic residence time ranges from minutes to days (Evangelidou et al., 2022). These four primary domains can be subdivided into intermediate reservoirs and sinks along land-to-sea continuum including atmosphere, catchment area, with lakes and soils, river systems, coastal systems, shelf to abyssal sediments, turbidite and drift systems and marine waters (Fig. 5).

Microplastics can be considered to enter intermediate reservoirs enroute to permanent storage within the sedimentary record. Given the importance of plastic pollution, the zones of temporary storage, or intermediate reservoirs, take on a greater significance when compared to sedimentary systems. One key area of temporary storage is the water column of bodies as lakes and sea. Due to the buoyancy of microplastic the ocean reservoir could be a key uncertainty in understanding the pathways from source-to-sink, as a large microplastics can get trapped within different water bodies due to the low settling rate of microplastics (e.g., Sonke et al., 2022). Given that a source-to-sink approach for microplastics would operate on relatively short time scales, decades or less, the oceanic reservoir, in particular, would be a region of important temporary storage, as are fluvial systems in the sedimentological S2S studies.

Plastic enters intermediate to permanent storage when it is deposited at the bottom of aquatic environments. Large fragments of plastic will get transported down rivers and most likely enter temporary storage within the riverbanks, channel bars and other morphologic features (Tramoy et al., 2020). Subsequently these fragments of plastic will either reach the river estuary and marine environment. Over time fragments of plastic will degrade and sink. It is estimated that nearly all plastic that has entered the ocean since 1950 has sunk below the ocean surface layer (Koelmans et al., 2017). This plastic sink is estimated to consist of 83.7% macroplastics, 13.8% microplastics and 2.5% nanoplastics based on mass budget approach (Koelmans et al., 2017).

Larger fragments of plastic, macroplastic, will initially float at the surface layer of the ocean and are impacted by different surface circulation processes from large scale gyres to Stokes drift from waves (van Sebille et al., 2020). Eventually, floating plastics are gathered into surface garbage patches by the ocean circulation main currents. It is worth noting that these garbage patches do not literally form islands of plastic but are surface waters with high concentration of plastic of different sizes. For large plastic particles to sink, two principal processes seem to play: (1) Biofouling, when organisms use the floating plastic as a refuge to grow leading to a denser object over time that starts to sink down the water column. (2) Larger fragments of macroplastic also enter directly into deep waters due to hyperpycnal flows, for example river floods, or collapse of bottom marine sediments, and continue to travel down slope along with gravity flows or turbidity flows (Tubau et al., 2015). Large fragments entering the deep sea without fragmentation have for example been found in submarine canyons in the northwestern South China Sea (Zhong and Peng, 2021).

A mass balance exercise via a box model for the cycling of plastic within the global environment would suggest that rate of exchange of microplastic from surface waters to the deep ocean is relatively slow, increasing longer-term storage within the subsurface waters (Sonke et al., 2022). Such dynamic corresponds to diffuse source for microplastics. Additionally, short-term event could lead to microplastic accumulation in distinct physiographic settings (e.g., Kane et al., 2020). They range from submarine canyons, which are a focus for gravity flows, and contourite drifts (accumulations of fine sediment built up by bottom ocean currents). Therefore, there is the potential that there are at least two general sources for microplastics that become sedimented out within the marine environment, (1) primary microplastics source, and (2) secondary microplastics coming from the degradation of large elements during the transport to the sink (see Table 1). Different pathways and associated accumulation sites could have different signature of these two types of microplastics, where the marine snow that blankets the abyssal plains and accumulates in contourite drifts may contain old microplastics via fragmentation, while canyon fans contain greater quantities of young microplastics that are sourced from point sources in the terrestrial environment.

## 5. Transport pathways

S2S studies typically rely on the transport pathways influenced by two main drivers: (i) wind and (ii) water in various forms, including ice, rivers, lakes, and marine environments (e.g., Einsele, 2000; Hinderer, 2012). Fig. 5 illustrates the various transport pathways (letters A to N in Fig. 5) that operate for the microplastic routing system and that are discussed in the following sections. Additionally, human activities play a significant role in transporting plastics throughout the environment (Fig. 5).

### 5.1. Atmospheric currents (A in Fig. 5)

Atmospheric transport of microplastics has emerged as a vector for the deposition of microplastics in both terrestrial and aquatic environments (e.g., Allen et al., 2019a; Allen et al., 2022a; Allen et al., 2022b). It has been demonstrated that atmospheric transport can carry

microplastics to remote areas without any local sources of plastics (Allen et al., 2019a). For example, microplastics have been detected on remote glaciers in the Alps and Tibetan regions, indicating that wind-driven transport may be responsible for their deposition in such high-altitude locations (Ambrosini et al., 2019; Zhang et al., 2019).

### 5.2. Rainfall and atmospheric fallout (B in Fig. 5)

Rainfall and snowfall events were then shown to directly contribute to the wet deposition of atmospheric microplastics (Dris et al., 2016; Allen et al., 2019a). Laboratory studies performed by Ganguly and Ariya (2019) have demonstrated that micro/nano plastics can serve as efficient cloud ice nuclei. This finding might also explain the observed slight correlations between microplastic counts and snow events reported by Allen et al. (2019b). Preliminary studies have established a relationship between the presence of microplastics in water bodies and rainfall events (Gündoğdu et al., 2018; Zhang et al., 2017). These findings suggest that rainfall events could distribute microplastics into terrestrial and aquatic environments, leading either to temporary storage in the soil or, if there is sufficient run-off, to direct transfer into the fluvial systems. Additionally, dry deposition, or the fall-out of microplastics from the atmosphere onto land or water surfaces, has also been reported as a mechanism of microplastic transfer from the atmosphere to the terrestrial environment (Allen et al., 2022b).

### 5.3. Surface runoff (C in Fig. 5)

Surface runoff (C in Fig. 5) is recognized as a significant pathway for transporting plastic debris from terrestrial environments into aquatic environments, especially in urban and agricultural areas (Siegfried et al., 2017; Piehl et al., 2021; Schernewski et al., 2021). However, despite its importance, the mobilization and transport processes for plastic via surface runoff remains poorly understood (Han et al., 2022). Various sources of microplastics have been recognized in urban runoff including industrial activities (Grbic et al., 2020), construction (Shruti et al., 2021), landfill leachate (Golwala et al., 2021; Sulistyowati et al., 2022), or the laundering of synthetic textile fabrics (Qian et al., 2021).

### 5.4. Fluvial transport and impact of vegetation (D in Fig. 5)

Sediment and microplastic transport modes in the river system environments are commonly divided into three categories: (i) surface transport, (ii) suspended transport, and (iii) bedload transport (Cowger et al., 2021; van Rijn, 1984a; van Rijn, 1984b; Ancey, 2020). (i) Surface transport involves the movement of larger, positively buoyant particles with high rising velocities. These particles resist downward turbulent mixing and are primarily found near the free surface of the water. (ii) Suspended particulate transport occurs when the settling or rising of particles through the flow field is counteracted by the turbulence of the flow itself. This phenomenon keeps the particles in suspension in the water column. (iii) Bedload transport is associated with the movement of larger particles that have high settling velocities relative to turbulent fluctuations in the flow. The bedload transport results in the sliding, rolling, and saltation of particles over the bed of the water body. Flood and overspill flows could also act as a major carrier for microplastic (e.g., Constant et al., 2020). Rivers are recognized as a major agent in transporting microplastics to coastal regions as they serve as conduits for microplastics from inland sources to the ocean (Moore et al., 2011; Klein et al., 2015; Horton et al., 2017; Lebreton et al., 2017; Hurley et al., 2018; Pierdomenico et al., 2019).

While literature on the effects of vegetation on sediment transport, there are still many unknowns regarding how vegetation mitigates the transport of microplastics (Zhang, 2017). Several studies have reported enhanced deposition of microplastics in both riverine and coastal ecosystems where various types of vegetation are present. These ecosystems include riparian vegetation and freshwater wetlands (Yin et al., 2021;

Ding et al., 2021), mangroves (Martin et al., 2020), salt marshes (Stead et al., 2020), and seagrasses (Kreitsberg et al., 2021; Sanchez-Vidal et al., 2021). Despite these observations, the specific mechanisms through which vegetation influences the transport and deposition of microplastics remain unclear. Further research is needed to understand the complex interactions between vegetation and microplastics.

#### 5.5. River mouth continuous discharge and river plume (E in Fig. 5)

Microplastic beaching rates have been found to depend strongly on the characteristics of the river mouth through continuous discharge (Atwood et al., 2019), and seashores downstream of river outflows tend to have higher densities of anthropogenic debris compared to seashores upstream or distant from river mouths (Rech et al., 2014; Cheung et al., 2016).

Coastal fronts, whether generated by river plumes, upwellings, or other processes, can act as barriers to the transport of floating items, including marine plastics (Hinojosa et al., 2011; Garden et al., 2014; Ourmieres et al., 2018). These fronts can trap and concentrate floating debris (van Sebille et al., 2020), including microplastics, leading to higher concentrations along the outer edges of river plumes (Atwood et al., 2019), as well as on the seafloor and shorelines (Acha et al., 2003).

Floods from rivers are highly episodic even with daily to yearly variability, and can deliver significant quantities of litter, including plastic, to the ocean (Moore et al., 2011; Rech et al., 2014, 2015).

#### 5.6. Coastline, beaching, and tide (F in Fig. 5)

The processes that control the beaching of plastic and other buoyant pollutants, such as oil, are still relatively understudied. Some attempts have been made to create data-driven estimates of litter on beaches using artificial neural networks based on cleanup survey data of large debris (Balas et al., 2004; Schulz and Matthies, 2014; Granado et al., 2019). Similarly, beach characteristics like steepness and sediment type could play a role in determining which plastic particles get stranded on beaches (Hardesty et al., 2017).

The hydrodynamics in coastal waters differ significantly from those in the open ocean, and they play a crucial role in controlling the transport of plastic particles. In the swash zone (the area of the beach where waves rush up and backwash down), plastic buoyancy and swash zone flow asymmetry are important factors determining the beaching of plastic (Hinata et al., 2017). Large floating particles located offshore but within a certain distance offshore are susceptible to beaching. Particles with low settling velocities are recaptured by small eddies induced by swash waves and transported seaward, while large particles with high settling velocities tend to remain in the swash region.

Plastic particles washed ashore during storm events or large wave events may deteriorate due to weathering and mechanical abrasion by sediments. As a result, they can fragment into smaller pieces that are then backwashed offshore by swash waves and wave-induced nearshore currents (Isobe et al., 2014; Kataoka and Hinata, 2015).

Coastlines are considered hotspots for microplastic generation, with plastic degradation being related to factors such as ultraviolet radiation, mechanical abrasion by sediments, and fragmentation in the sea swash and wave breaking zone, especially during storm events (Chubarenko and Stepanova, 2017; Chubarenko et al., 2018; Efimova et al., 2018). The rate of fragmentation of beached plastic is closely related to the residence time on the beach and is dependent on the type of polymer and temperature conditions (Song et al., 2017; Andrady, 2011).

It has been also reported that uncontrolled burning of in-situ plastic wastes on beaches can generate a new material referred to as “plastic-glomerates”, mainly containing melted plastics mixed with beach sediments and organic debris (e.g., Corcoran et al., 2014; Utami et al., 2023).

#### 5.7. Coastal current and longshore drift (G in Fig. 5)

Buoyant plastic particles and macroplastics, due to their tendency to remain at the sea surface, are more influenced by onshore drift, which carries them towards the shore. In contrast, smaller plastic particles are more likely to be mixed into the water column and follow the undertow, which is the offshore flow of water returning to the sea after waves break on the shore (Isobe et al., 2014; Kataoka and Hinata, 2015). As a result, microplastic particles may be transported away from the shoreline and farther offshore.

Observational surveys in the southern Mediterranean have shown that more floating plastic bottles are found inside the surf zone (the area where waves break near the shoreline) than outside it, supporting the idea that buoyant plastic particles tend to accumulate closer to the shore (Pasternak et al., 2018).

As of now, the specific impact of longshore drift on microplastic distribution has not been conclusively demonstrated in scientific literature. However, due to its similar properties to fine grain sediment or organic material, it is highly plausible that longshore drift plays a significant role in the transport and distribution of microplastics in coastal environments (Kane and Clare, 2019).

#### 5.8. Downwelling and upwelling (H in Fig. 5)

Both upwelling and downwelling are typically initiated by wind-driven processes or large-scale ocean currents like Ekman currents (Fig. 5). These processes play a crucial role in shaping the distribution and dynamics of organic matter in marine environments, especially along coastlines and continental shelves. Despite their importance, there are limited studies specifically dedicated to understanding the influence of upwelling and downwelling on microplastic transport in the marine realm (e.g., Choy et al., 2019; Díez-Minguito et al., 2020).

#### 5.9. Turbidity and gravity currents (I in Fig. 5)

Turbidity currents (I in Fig. 5) also play a crucial role in the transport of both microplastics and sediments from the shelf down to the deep sea (Kane and Clare, 2019; Pohl et al., 2020; Harris et al., 2023). When sediment from a river system enters an open water body, it can carry both natural and plastic particles in two different types of flows: hyperpycnal (denser flow moving beneath the basin water) and hypopycnal (less dense flow moving above the basin water). Plastic particles can be transported by hyperpycnal flows, typically generated by flash floods (Pierdomenico et al., 2019) or deep-water cascading events (Sanchez-Vidal et al., 2015; Tubau et al., 2015).

Laboratory flume experiment also revealed that turbidity currents carrying microplastic particles and fibers led to a deposit enriched in microfibers (Pohl et al., 2020). This suggests that microfibers were trapped by settling sand grains at the base of the flow, while microplastic particles were transported and ultimately preserved on top of the resulting bed (Kane and Clare, 2019; Pohl et al., 2020).

Submarine canyon cutting the continental slope can serve as long-term pathways for the transport and deposition of microplastics into the sea through gravity flows. These gravity-driven flows, such as turbidity currents or sediment-laden flows, often decrease in energy both spatially and temporally as they move downslope along the canyon. Lower density and finer-grained particles, including microplastics, can be transported farther by the currents compared to higher density and coarser-grained particles (Kane and Fildani, 2021). These processes have been observed in the transport and deposition of low-density particulate organic carbon, as studied by Zavala et al. (2012), McArthur et al. (2017), and Paull et al. (2018).

Microplastics, initially transported by turbidity currents, can be carried further by dense thermohaline bottom currents (L in Fig. 5, see 5.12 below), resulting in contourite deposits (Kane et al., 2020).

### 5.10. Open ocean Stokes drift and Langmuir circulation (*J* in Fig. 5)

The motion of a particle floating on the free surface of a gravity wave experiences a net drift velocity in the direction of wave propagation, known as the Stokes drift (Stokes, 1847). It is still unclear how floating plastic particles are transported according to their shapes, densities, sizes, and the velocity of the surrounding marine currents including the Stokes drift (van Sebille et al., 2020).

Early studies focused on debris accumulation near Hawaii (Kubota, 1994), found that Stokes drift derived directly from local wind fields excluding swell did not significantly contribute to debris transport. However, more recent modeling studies considered Stokes drift from the entire wave field, combining wind and swell waves, and found a greater role for Stokes drift in the transport of marine plastic particles (Iwasaki et al., 2017; Delandmeter and van Sebille, 2019; Fraser et al., 2018; Dobler et al., 2019).

On a global scale, Stokes drift does not significantly contribute to large-scale accumulation of microplastics in the subtropics, but it does lead to increased transport to polar regions where storm-generated waves are larger and more frequent (Onink et al., 2019). Instead, the formation of Langmuir cells, resulting from the interaction between wind-induced shear flow and wave-induced Stokes drift, further influences the horizontal dispersion of buoyant materials in the ocean surface layer (Colbo and Li, 1999; Kukulka and Veron, 2019).

### 5.11. Large-scale open ocean processes (*K* in Fig. 5)

While the Ekman/geostrophy theory has been valuable in explaining the large-scale distribution of floating microplastics in the ocean and the formation of garbage patches and gyres (Kubota, 1994; Martinez et al., 2009; Onink et al., 2019), the complete understanding of the pathways and dynamics plastic particles requires consideration of additional small-scale features, such as eddies and striations that are able to modulate the transport of surface material (e.g., Goldstein et al., 2012; Maes et al., 2016; Brach et al., 2018). Further research and interdisciplinary efforts are needed to unravel the complexity of plastic transport and accumulation in marine environments. Specially as most models do not consider actual pathways and their real time scales of the different plastic particles from their sources down to the open ocean accumulation zones (van Sebille et al., 2020).

### 5.12. Contourite, bottom current and deepwater drift (*L* in Fig. 5)

The settling of microplastics to the seabed is influenced by the balance between shear stress at the base of the flow and the settling velocity of the microplastics. When the shear stress is lower than the settling velocity, settling is inhibited, and microplastics can be advected by the flow. In many worldwide deep-sea locations, the typical velocities of near-bed thermohaline currents range from 0.1 to 0.4 m/s (McCave et al., 2017), which controls microplastic transport in deep-ocean locations (Fischer et al., 2015; Bergmann et al., 2017; Peng et al., 2018), leading to the formation of extensive seafloor contourite deposits (Kane et al., 2020).

### 5.13. Vertical transport and mixing (*M* in Fig. 5)

The vertical distribution of floating plastics in the ocean is influenced not only by the buoyancy of the particles but also by the dynamic pressure resulting from vertical movements of ocean waters. This dynamic pressure can cause plastic particles to move up and down in the water column (van Sebille et al., 2020).

Additionally, plastic particles may be dragged by or adhere to marine snow, which is a type of organic material mainly composed by phytoplankton, bacteria, fecal pellets, live protozoa and zooplankton and biodebris that sinks through the water column. This process leads to the formation of sinking aggregates (flocs) that contain both marine snow

and microplastic. Regional observations have effectively shown that a significant proportion of marine snow aggregates, up to 70% in some cases, can include microplastics (Zhao et al., 2018; de Haan et al., 2019).

### 5.14. Biota and organisms (*N* in Fig. 5)

Organisms can play a significant role in the transport of plastic debris in the marine environment. Birds have been documented to transport (ingestion or entanglement) plastic particles (e.g., Buxton et al., 2013; Le Guen et al., 2020) and then redistribute plastic along long distance migration routes (e.g., Rummel et al., 2016).

In addition, plastic particles can be incorporated into sinking fecal pellets by marine organisms. Plankton, fishes, seabirds, and marine mammals have been documented to ingest and pack microplastic into fecal pellets (Lee et al., 2013; Cole et al., 2015, 2016; Katija et al., 2017), which significantly influence the sinking rate of fecal pellets (Cole et al., 2016). The involvement of organisms in the transport of plastic debris adds yet another layer of complexity to the dynamics of plastic pollution in the ocean.

## 6. Source-to-Sink microplastic mass balance

### 6.1. Balancing the budget of the environmental plastic cycle

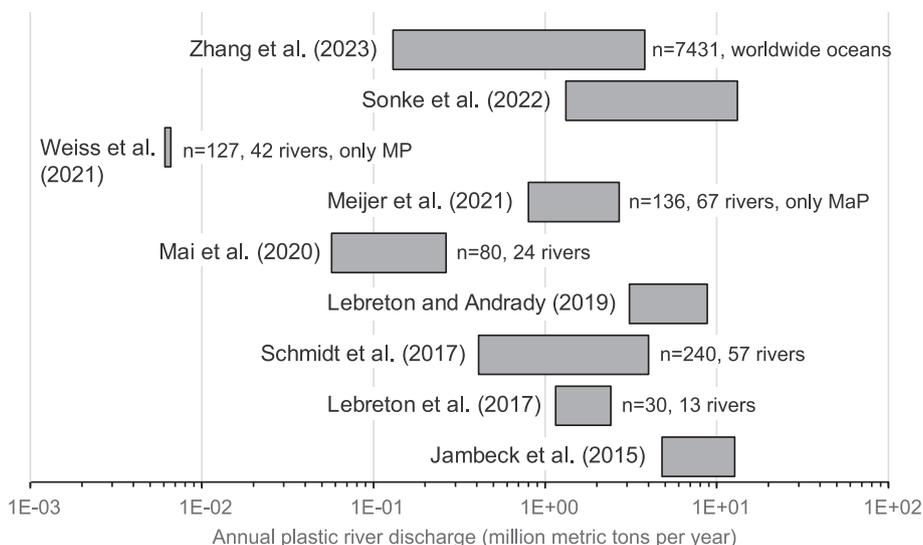
While some plastics are lost or deliberately discarded from ships, most enter the marine environment from the rivers. Several modeling studies have described and estimated the amount of plastic in the ocean (Fig. 6) (Jambeck et al., 2015; Lebreton et al., 2017; Schmidt et al., 2017; Mai et al., 2020; Weiss et al., 2021; Sonke et al., 2022; Zhang et al., 2023). Overall, considering the maximum value of 14 million ton/year, accounting for around 3% of all plastic production (Jambeck et al., 2015), the total plastic flux can be compared to the estimated 12.5 billion ton/year of sediment delivered by rivers to the coastal marine environment (Syvitski et al., 2005). The mass of sediments entering the oceans annually is thus about ~900 times greater than the maximum mass of microplastic particles.

Comparatively, the total load of particulate organic matter (POM) reaching the oceans from all rivers is estimated to be around 200 million ton/year (Hedges et al., 1997), which is approximately 15 times greater than the maximum estimated flux of plastic mass (Fig. 6). In other words, the current maximum plastic load in the ocean accounts for about 7% of the POM entering to the marine environment. These comparisons highlight the substantial difference in magnitude between the mass of plastic particles and the mass of sediments and particulate organic matter transported by rivers to the marine environment.

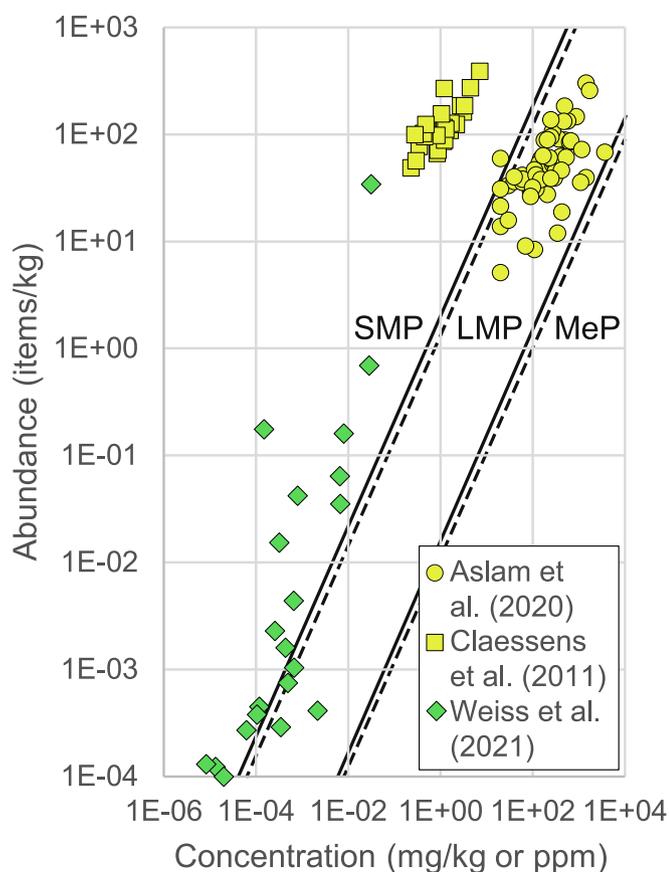
The variety of methods used to measure plastic and microplastic concentrations in water and sediments has produced results that are difficult to compare (Fig. 6) and which has hampered attempts to draw general conclusions about the fate of plastic in the environment. For instance, Jambeck et al. (2015) calculated that 4.8–12.7 million tons (Mt) of plastic entered the oceans from land-based sources in 2010. Lebreton et al. (2017) reported that from 1.15 to 2.41 Mt. of plastic waste enters to the ocean from rivers every year, while Schmidt et al. (2017) estimated an inflow to the oceans of 0.47–2.75 Mt. per year.

Weiss et al. (2021) dedicated significant efforts to address the influence of the empirical relationship used to convert the number of microplastics per sampled volume into mass of microplastics per sampled volume. They compared the impact of this conversion on the works of Lebreton et al. (2017) and Schmidt et al. (2017). Their findings revealed that the correction proposed by the Weiss factor (Wf) led to a substantial minoring of the estimated tons of microplastics at the river outlet, surpassing an order of magnitude (Fig. 6).

This highlights the critical importance of the regression law used to convert the number of microplastics (abundance) into the mass estimation (concentration). The shape as well as the distribution of microplastics add on the difficulty to make such conversion (Fig. 7) (Leusch



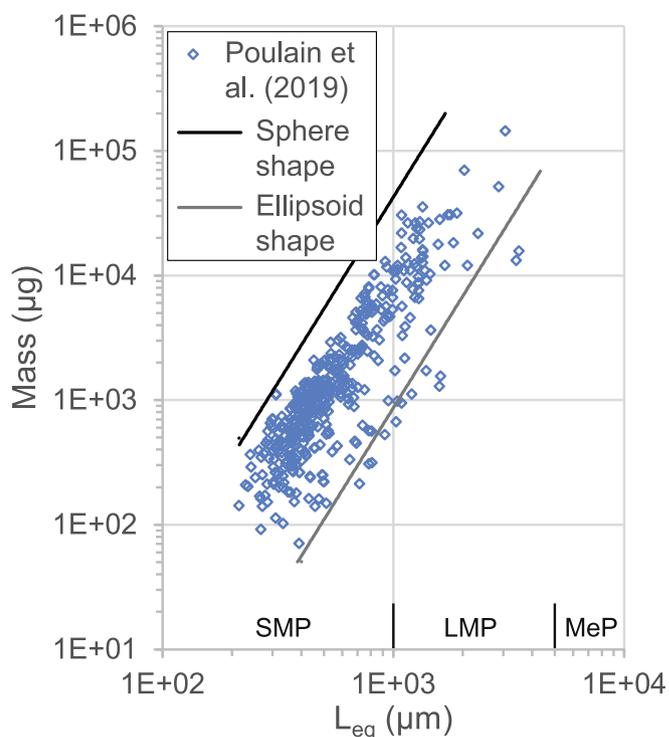
**Fig. 6.** Comparison of the main studies to estimate global plastic riverine discharge in millions metric tons per year. (1) High: riverine emissions as suggested by Lebreton et al. (2017); (2) Middle: riverine emissions by Mai et al. (2020); and (3) Low: riverine emissions by Weiss et al. (2021). The Lebreton et al. and Mai et al. inventories are for total plastics, including both microplastics (diameter < 5 mm) and macroplastics (diameter > 5 mm), while the Weiss et al. inventory only includes microplastics.



**Fig. 7.** Mass to number relationship, or Abundance versus Concentration relationship, showing the influence of the particle grain size on the order of magnitude of the resulting estimates (data from Claessens et al., 2011; Aslam et al., 2020; Weiss et al., 2021 and references herein). Regression lines subdividing the graph into small microplastic (SMP, <1 mm), large microplastic (LMP, 1–5 mm) and mesoplastic (MeP > 5 mm) are derived from Leusch and Ziajahromi (2021). Dotted black line is computed for PVC (density = 1.38 g/cm<sup>3</sup>). Continuous black line is computed for PE (density = 0.94 g/cm<sup>3</sup>).

and Ziajahromi, 2021). Alternatively, relationships depending on the size-scale of each individual particle could also be used to estimate microplastic masses (Fig. 8).

Particle size seems a crucial parameter for making robust conversions between the number and mass of microplastics (Fig. 8). However, measuring the length of microplastic particles can be analytically challenging, especially when dealing with many particles that are also very small. Automated imaging systems and microscopy techniques can be



**Fig. 8.** Length to mass relationship (data and regression lines from Poulain et al., 2019). Automatic dimension to mass conversion is for example proposed into the post processing software tool (siMPle) for spectroscopic analysis presented in Primpke et al. (2020).

employed to measure particle lengths efficiently (Cowger et al., 2020). The complexity and variability in shapes and densities of microplastics are such that uncertainties in the estimations will likely remain and lead to orders of magnitude variations in the results for the total mass of plastic in both sources and sinks.

This aspect is of utmost importance in establishing accurate mass balances for microplastics. The challenge lies not only in establishing a universal relationship but also in the scarcity of data to constrain these empirical laws. For instance, the synthesis on microplastic sinks in the marine domain conducted by Harris (2020) showed that only two (Claessens et al., 2011; Alsam et al., 2020) out of 80 analyzed articles, provided sufficient data to establish a relationship (Fig. 7). A similar scarcity of data is observed in the oceanic domain, as seen in the synthesis proposed by Harris et al. (2023), where out of 23 studied articles, only one allows for a discussion of such a relationship (Jones et al., 2022).

This historical issue stems from the initial methods used to quantify plastic pollution in the environment. Visual counting, which was the simplest method available since the 1970s (Carpenter et al., 1972), served as a reference until the 2010s when thermal methods including pyrolysis-gas chromatography–mass spectrometry (Py-GC–MS), thermogravimetry (TGA), and differential scanning calorimetry (DSC) as well as other advanced analytical techniques (Rock-Eval® device), were introduced to quantify microplastics in environmental samples (e.g., Peñalver et al., 2020 and references therein; Romero-Sarmiento et al., 2022). These modern methods enabled direct mass quantification. However, visual counting or other quantification methods ( $\mu$ FTIR, Raman, among others) still need to be improved, to refine the regression that establishes the relationship between the number and mass of microplastics per investigated volume.

Another significant challenge in conducting mass balance studies for microplastics is the lack of information on the deposition duration associated with a given concentration. Without knowing the duration

over which a certain concentration is present, it becomes difficult to accurately convert concentration data into flux data (Lebreton et al., 2019). To establish comprehensive global mass balance for macroplastics and microplastics, it is crucial to consider the baseline level of plastics that already exists in the environment before any additional deposition or transfer occurs. This baseline needs to be established before conducting any measurements, and then the subsequent measurements are compared to the reference value during a specified time interval.

In addition, fragmentation of macro- or microplastics into nanoplastic should not be neglected particularly, as fragmentation is accelerated by many environmental factors. The most important are UV radiation and mechanical wear or stress (Pinto da Costa et al., 2016). However, microbiological activity and other biological factors (e.g., digestive fragmentation, Dawson et al., 2018) can enhance plastic debris fragmentation in the environment. Pfohl et al. (2022) state that the half-life of e.g., thermoplastic polyurethane particles (100  $\mu$ m, considering mid-European climate conditions) until the particle becomes water-soluble organics and fragments is about 73 years. Further studies are still needed to better evaluate these processes.

## 6.2. Plastic concentration from source-to-sink

To assess how microplastics accumulate from source-to-sink, it was necessary to focus on publications that provide mass concentrations to minimize uncertainties associated with the number-mass conversion (Fig. 9). However, it is important to note that other uncertainties can arise from various steps of the evaluation. Samples preparation, volume representativity, and finally the identification and quantification methods with various limits of detection that can impact the measured concentrations. Given these challenges and uncertainties, it is important to interpret the presented synthesis results (Fig. 9) with caution although it is based on the best available information to date. Further

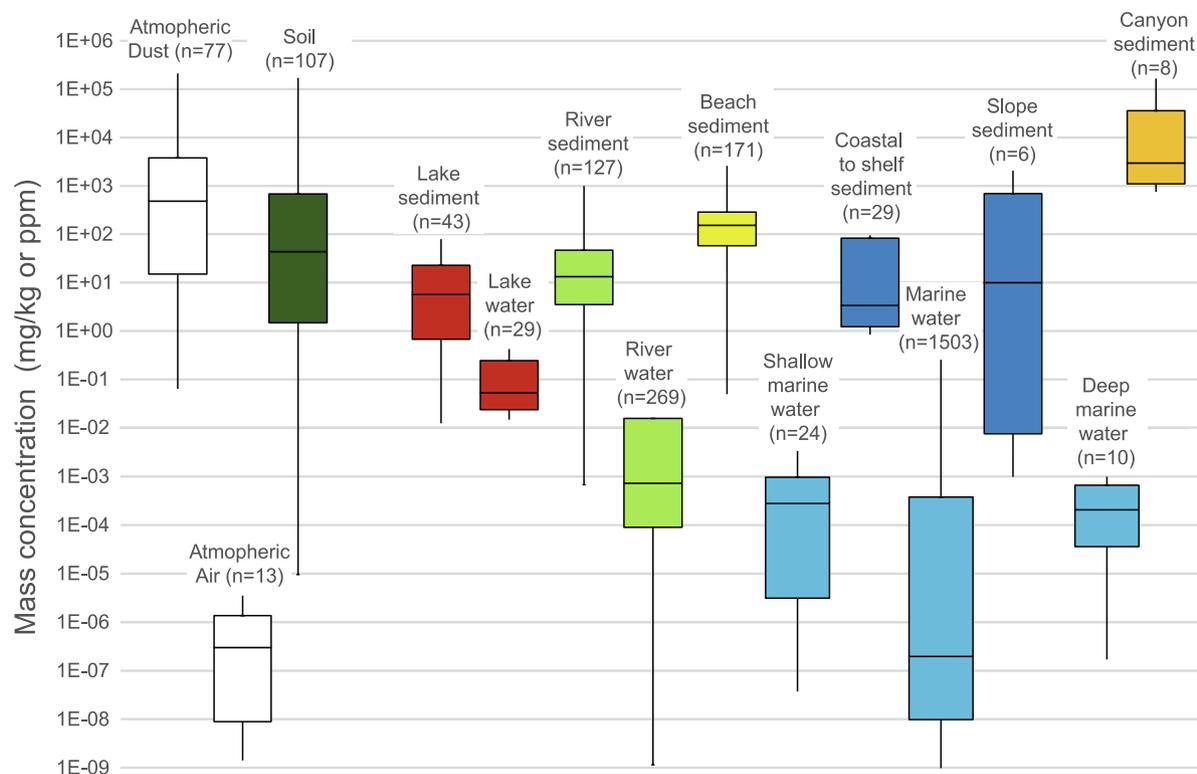


Fig. 9. Representation of the volumetric mass concentration (mg/kg or ppm) of microplastics in box plot along the source-to-sink profile including sedimentary, air and water matrices. The color code is the same used in Fig. 5. Minimum and maximum values, as well as the first and third quartiles and the median are represented for each box plot. (n) indicates the number of measurements for each box. References used and dataset are listed in Table 2.

research and additional works are essential to refine our understanding of microplastic accumulations from sources to sinks.

As stated above, the objective of this review was to determine mass concentration values per volume investigated from source-to-sink, encompassing surface water, marine waters, sediment, atmosphere, air, and fallout deposits. The exercise proved to be challenging as the available references with published mass data were limited, particularly for the deep-sea domain (Fig. 9). By focusing on the trends, our study emphasizes the broader patterns while avoiding excessive detail about absolute concentration values.

A total of 1503 measurements were found for marine waters, forming the most abundant dataset. For the sedimentary part, eight compartments were documented, including atmospheric dusts, soils, lake sediments, river sediments, beach sediments, coastal and continental shelf sediments, continental slope sediments, and sediments in submarine canyon (Fig. 9). Unfortunately, no mass data references were found for the deepest part of the land-sea continuum.

To gain insights into the distribution of mass concentrations of microplastics, data are presented in the form of box plots which allows to illustrate, at once, minimum and maximum and the median values, as well as first and third quartiles (Fig. 9). Based on the trends observed in the median, first, and third quartiles, an initial description of the distribution of mass concentrations of microplastics can be provided.

Concentration values in atmospheric deposits are very high, but very low in the atmosphere samples (Fig. 9). In the deposits along the continental-ocean continuum, no clear trend can be found. Some compartments show very high variability, it seems that maximum values are found in marine canyons, soils and on the beaches, whereas the minimum are measured in lake deposits (Fig. 9).

These concentrations need to be considered with caution due to sampling: some areas are under-sampled (only 8 samples from canyons), others might be biased by the purpose and chosen study sites. In any case, concentrations in sediments are several orders of magnitude higher than concentrations in waters which show a clear upstream to downstream decreasing trend (Fig. 9).

Lake waters demonstrate maximum concentrations, while marine waters have minimum concentrations, which are 1 to 6 orders of magnitude lower than those in deposits (Fig. 9). Lacustrine waters show the least difference between sediment and water concentrations. Microplastics seem to concentrate at interfaces within marine waters (shallow and deepest part) and beaches at the land-sea interface. A similar pattern is observed for soils and atmospheric deposits at the land-atmosphere interface. Larger and more dynamic water masses show lower concentrations (e.g., marine domain), like the dilution effect observed in the atmosphere, where atmospheric concentrations are extremely low compared to indoor environments (Dris et al., 2016). Soil

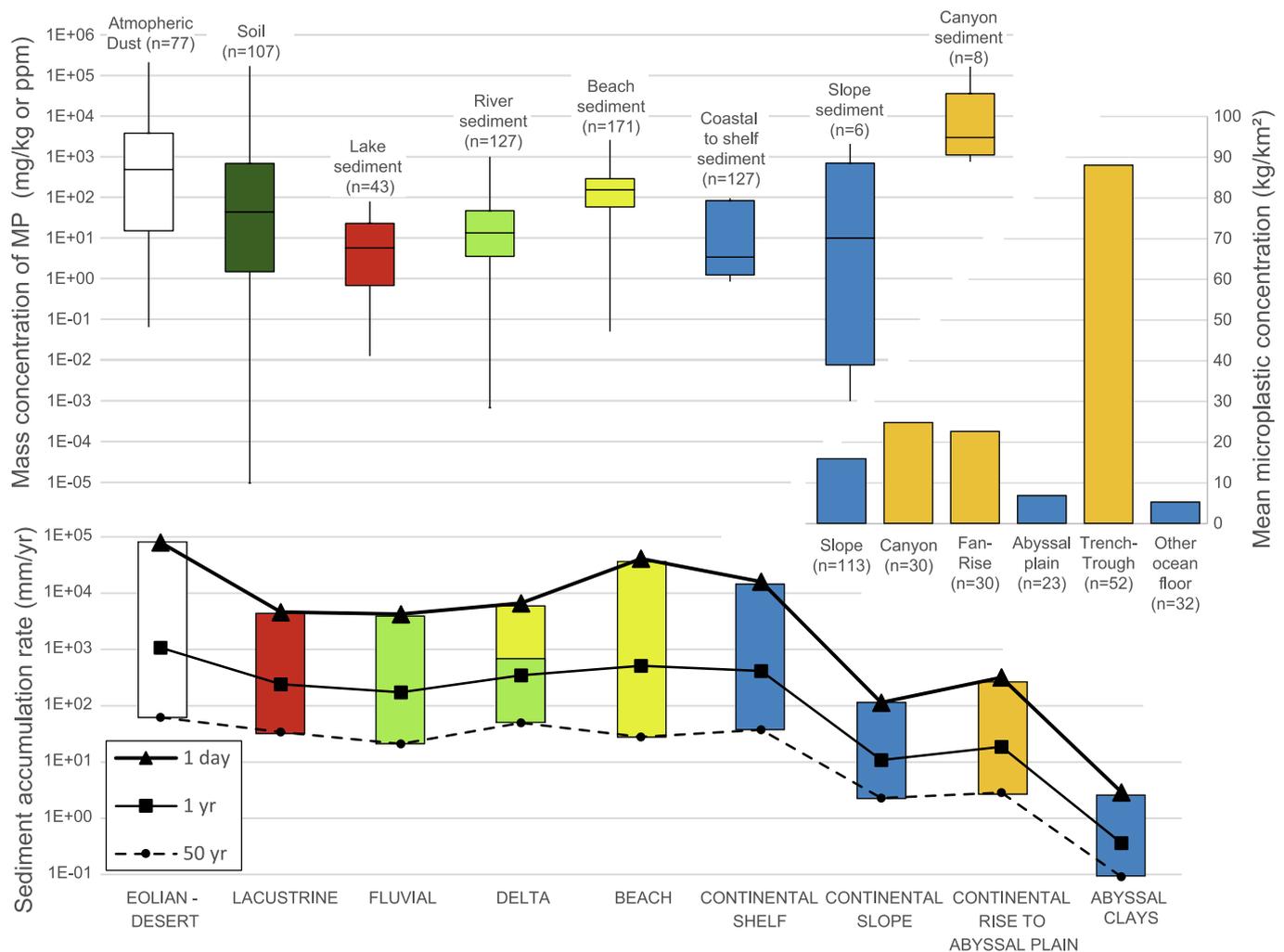


Fig. 10. Plastic volumetric mass concentration (mg/kg) for sedimentary matrices (top) versus sediment accumulation rates (base) across the land-sea continuum. Sediment accumulation rates (mm.yr) are represented for 1 day, 1 year and 50 year of observation. Color code are the same used in Figs. 3 and 6. Minimum and maximum values, as well as the first and third quartiles and the median are represented for each box plot. (n) indicates the number of measurements for each box. Mean microplastic concentration for deep-water deposits (right) are from Harris et al. (2023).

concentrations are notably high, possibly due to the low flux of natural particles inducing a relative abundance of microplastics (Fig. 9). Proximity of numerous sources are also accounted for such high values.

### 6.3. Dilution, hot spot and final sink?

To gain further insights into the distribution of microplastic concentrations in depositional environments, a review of sedimentation rates was performed (Fig. 10) to identify whether a dilution phenomenon exists and how sedimentation processes might influence microplastic concentrations. Long-term deposition is made up of the accumulation of individual bedforms from event that led to the accumulation of both sediment and plastics. Stratigraphers have recognized the stochastic nature of these accumulations, and the importance of durations of deposition, erosion, and non-deposition that can skew the apparent sedimentation rates (e.g., Sadler, 1981). Over a short timescale, the record of deposits will be biased towards events with a low residence time, while as the timescale increases, the record is biased towards deposits that are preserved. The extensive database of Sadler (1981) gathering representative sedimentation rates over specific observation time steps was used here (Fig. 10) to address the potential bias in synthesizing sedimentation rates caused by the timescale of observation.

By analyzing different observation time steps (50 years, 1 year, and 1 day) sedimentation rates were estimated that are representative of the durations relevant to studying microplastic distributions in the environment (Fig. 10). Generally, that analysis shows a two-order-of-magnitude difference in sedimentation rates between an observation time step of 1 day versus 50 years. An interesting observation emerged when comparing microplastic concentration trends with high and low values against sedimentation rates (Fig. 10). Our analysis suggests that there is no dilution process at play. Instead, it appears that microplastics are captured and concentrated within the sedimentation process. Notably, environments with faster sedimentation rates exhibit a higher potential to concentrate microplastics (Fig. 10).

While the relationship between sedimentation rates and microplastic concentrations seems evident, questions arise regarding the availability of microplastics before deposition in different environments. The deposition processes vary significantly between environments such as beaches and submarine canyons, which may influence microplastic availability and accumulation.

Our analysis reveals that sediments deposited in submarine canyons exhibit higher microplastic concentrations compared to beaches and rivers. Interestingly, the gravity currents passing through these canyons transport sediments from these intermediate reservoirs like beaches and rivers, further illustrating the ability of turbidity currents to concentrate microplastics in deposits.

To gain deeper insights into microplastic distribution in poorly sampled distal and deep-sea areas, data synthesized by Harris et al. (2023) were also integrated. These data were not considered in the previous synthesis (Fig. 9) due to their mass concentration units reported by surface and not volume (Fig. 10). The addition of these data seems to reinforce the evidence of turbiditic and contouritic systems' ability to over-concentrate microplastics. These deep-water systems emerge as important hotspots, with concentrations surpassing even those found in soils, which are already known for their high microplastic contents.

Turbiditic systems, located at the end of the source-to-sink profile, or land-sea continuum chain, do not function as intermediate reservoirs like rivers or beaches. Instead, they represent an ultimate hotspot sink for microplastics, potentially interacting with associated contouritic systems.

## 7. Way forward

### 7.1. Limits and requirements

#### 7.1.1. Counting versus mass quantification

The comparison of plastic mass estimations from riverine input (Fig. 6) clearly illustrates the importance of number-mass conversion relationship for a given volume. The proposed synthesis in Fig. 9 illustrates that very few data are available to establish such relationships. Currently, there is no single analysis method that can provide both abundance and mass concentration results for microplastics. Therefore, combining multiple methods is necessary to obtain both pieces of information.

Many sampling approaches for microplastics in fluvial settings have been adapted from well-established techniques used in marine environments. Several factors, including flow regime and discharges, can influence the effectiveness and relevance of these sampling methods in continental settings (Li et al., 2018). Given the low concentrations of microplastics in many environments and the limits of detection of analytical methods, Koelmans et al. (2019) recommend a minimum sampling volume of 500 l for river water, regardless of the chosen sampling approach. Such recommendation should be extended to the different matrices and deposition environments to facilitate comparison between measurements.

The diversity of both sediments and microplastics, coupled with the wide range of existing methodologies, should compel the scientific community to establish consistent and comparable sampling and quantification approaches for future research. These guidelines should encompass the entire assessment process of microplastics in environmental samples, from sampling techniques to samples preparation and the identification of microplastic particles, but also some additional transdisciplinary approaches to intercompare results.

Proposals to refine the classification of microplastics should incorporate the establishment of subclasses that account for both the methods of sampling and analysis. Additionally, considerations regarding the dynamics of transport should be integrated in such proposals, potentially drawing inspiration from sedimentological expertise that considers particle size and interactions. For example, defining a subclass for very small microplastics (VSMP) with an upper limit at 70  $\mu\text{m}$  (ca. silt – sand limit) could be a valuable approach, especially considering feedback from cohesive and non-cohesive sediment behaviors.

#### 7.1.2. Microplastic particle descriptor and their dynamic properties

The wide diversity of microplastics, in terms of shape, size, and density, represents a singular challenge in comparison to natural sediments, which generally can be described by homogeneous spheres of different sizes. As a result, traditional sedimentary research approaches may only be directly applicable to a small range of the microplastics particles to understand their behavior and their fate in the environment. Dataset gathered by Poulain et al. (2019) suggests that environmental microplastic could be represented with sphere to ellipsoid shapes (Fig. 8).

For shape and size, researchers should improve and standardize the physical description of microplastic particles further than a diameter and an anisotropy factor. Microplastics, defined by their size range of 1–5000  $\mu\text{m}$  in diameter, cover a broad spectrum of sedimentary particulate sizes including clay, silt, sand, and even small gravel particulates (Fig. 1). These sediments exhibit diverse origins, transportation mechanisms, sedimentation properties, and interactions with other constituents like organic matter. Similarly, microplastics likely exhibit variability in these aspects, necessitating a revision and standardization of the grain size scale and nomenclature for plastic-to-microplastic classification, specially to support research in transport dynamic processes.

Moreover, changes in shape over time, driven by processes like fragmentation, degradation, aggregation, and deformability, adds

complexity to their behavior and predictability. These processes have received less attention in classical sedimentary research but seem to be crucial in understanding transport and fate of microplastics in the different environmental settings. As a result, microplastics can be viewed as a unique mineralogical-organic complex particle that requires further investigation to comprehensively assess its role and distribution in sedimentary systems.

### 7.1.3. Time and space scaling when monitoring microplastics

In river systems, upscaling from daily discharge measurements to yearly discharge estimation and their comparison to other parameters such as sediment supply from bedrock erosion is a significant challenge (e.g., Turowski, 2021). On the other hand, downscaling from monthly to daily scales is also challenging and may lead to significant underestimation of the flow event peak parameters. Sophisticated numerical modeling and data analysis techniques are necessary to accurately travel across time scales (e.g., Reborá et al., 2016) but also space scale. Such methods should be further investigated with more data to establish generic macroscopic descriptors for replicable mass balance studies and trustworthy correlation relationships between particulate organic matter and microplastic concentrations in the environment (Bergmann et al., 2017; Tekman et al., 2020).

Proper time and space scaling is also crucial for solid matrices of natural sediments. Sadler (1981) highlighted the significant changes in sedimentation rates observed over different time scales (see also Fig. 10). This emphasizes the importance of understanding and quantifying sedimentary processes (including hiatus periods) to differentiate short-term event deposits from longer-term sedimentation which are crucial to understand accumulations of microplastics in the different environment compartments.

### 7.1.4. From concentration to fluxes

To convert microplastic concentrations into microplastic fluxes, several approaches have been proposed and commonly required transfer coefficients (e.g., Sonke et al., 2022). They fall into two categories listed hereafter: (i) a simple approach, which compares plastic flux estimates with masses in upstream and downstream intermediate reservoirs, and (ii) a theoretical/experimental approach directly based on measured rates, such as sedimentation rates. For instance, the first category involves measuring microplastic flows, moving on a given surface of a river over a specific time. Comparison of these fluxes to the masses of plastic present in the upstream and downstream reservoirs allows the estimation of transfer coefficients. Sonke et al. (2022) introduced a box model approach covering the entire land-sea continuum, including atmosphere interactions, that is representative for the second category. They used a synthesis of observation data from 2005 to 2022 as well as model results from Brahney et al. (2021) for the atmosphere to estimate the transfer coefficients needed to close the global mass balance. Then, they were able to quantify fluxes. This comprehensive model includes macroplastics, large microplastics (LMP), and small microplastics (SMP), and represents a significant breakthrough on understanding plastic transfers in the environment.

The work performed by Sonke et al. (2022) is consistent with estimates proposed by Jambeck et al. (2015) and Lebreton et al. (2017) regarding flows at river outlets. However, significant discrepancies exist with Weiss et al. (2021), who proposed significantly lower values for microplastics. This raises critical questions about mass balance completeness and the conversion from plastic concentrations to plastic fluxes both for macroplastic and microplastic distributions in the environment. The need for a comprehensive grain size scale that accurately represents the diversity of microplastics, together with an updated nomenclature, is becoming increasingly apparent. While the box model approach represents a significant step forward, uncertainties remain substantial due to limited data available over all the compartments and questions on the quantification methods and the representativity of measurements. It highlights the need for further research and additional

observations in this field.

## 7.2. Insight from S2S and stratigraphy for plastic cycle studies

Source-to-sink approaches can play a crucial role in enhancing our understanding of the plastic cycle in the environment. As mentioned earlier, such approaches can provide valuable insights into the transfer coefficient or sedimentation rate estimates, which are essential for assessing the transport and fate of plastic particles within natural systems. Additionally, employing source-to-sink approaches allows for a comprehensive and holistic understanding of the plastic cycle, aiding in the improvement of waste management strategies, and the coordination of effective solutions to tackle plastic pollution and its environmental impacts.

### 7.2.1. S2S for improving waste management and reduce plastic usage

To effectively mitigate environmental impacts of mismanaged plastic waste, it is essential to have an accurate knowledge of both the distribution and the concentrations of plastics in the environment. Monitoring the efficiency of mitigation efforts also relies on this information.

In this context, Nyberg et al. (2023) proposed a S2S approach to explore the exposure of various river systems to MPW and make future predictions based on different plastic usage and mitigation scenarios. This study evaluates current and potential future impacts of plastic pollution in river systems under various management scenarios. It provided valuable insights for policymaking and effective plastic waste management. Combined with a circular economy approach, it could help account for the full environmental costs and benefits of plastic use and disposal (Hoang et al., 2022).

### 7.2.2. Modeling tools for risk assessment and mitigation coordination

Laboratory studies indicate that microplastics capability to absorb contaminants equals or surpasses to soils or sediment particles (e.g., Wang and Wang, 2018; Klöckner et al., 2021). Additionally, microplastics are known to be a major health issue. Considering the role of microplastics as chemical vectors and pollutants in the context of other potential exposure routes is essential for a thorough risk assessment of plastic debris contamination.

For assessing the risks of plastic debris on wildlife, it is important to understand the extent of its exposure. This involves studying the pathways through which plastic debris is transported from its sources (e.g., coastal areas, rivers, land-based sources) to marine environments (e.g., Hardesty and Wilcox, 2017; Everaert et al., 2018; Compa et al., 2019).

Monitoring and real-time data are required to reach an accurate knowledge on how plastics are transported (Kataoka and Hinata, 2015; Sherman and van Sebille, 2016; De Frond et al., 2019). Then, this information can be used to validate and to update numerical models on how plastic pollution spreads and accumulates in different regions, in order to enhance their predictive capabilities, provide more accurate assessments of plastic distributions in the environment, and guide strategies for mitigation.

## 7.3. How plastic could support S2S studies and stratigraphy?

Several key points currently emerged regarding the value of studying plastics to make insights for source-to-sink studies and stratigraphy that will be detailed in the following sections, with (i) plastics as tracers, (ii) impact of plastics on climate change (iii) plastics as a marker of the Anthropocene, and (iv) understanding fluid physics and aggregation.

### 7.3.1. Source tracer and transport processes

Plastic particles could serve as a unique tracer due to their solid nature and direct association with human activity. Plastics originating from various sources could be used as a powerful tool to study and understand physical oceanographic phenomena across various spatial and temporal scales (van Sebille et al., 2020). It could then result in valuable

insights into oceanic processes such as currents, eddies, and mixing at different scales. In river systems, these plastics could also be used to address residence time in fluvial systems or dynamics of intermediate reservoir storage.

The importance of future analysis to better characterize microplastic concentrations in poorly sampled areas cannot be overstated. A focus on deep water systems should be addressed to complete global mass balances and global microplastic distributions. It will of course give some new insights on gravity current processes as well as on the interaction between turbidite and contourite that is currently a hot topic in sedimentology and stratigraphy.

### 7.3.2. Impact of microplastics on climate

Recent studies highlighted that atmospheric microplastic and nanoplastic particles can also affect climate through theoretical influences on surface albedo (Brahney et al., 2021), cloud formation (Ganguly and Ariya, 2019) and radiative forcing (Revell et al., 2021). Considering the projected increase microplastic and nanoplastic particles in the atmosphere due to the rising global plastic use and mismanaged plastic waste, there is a concern that microplastic effective radiative forcing will also increase in the future. This may have implications for local and regional climates, especially in areas where airborne plastic concentrations are particularly high (Allen et al., 2022a). Additionally, as anthropogenic aerosol emissions are expected to decrease in the future, the relative contribution of microplastics and nanoplastics to total aerosol effective radiative forcing is likely to become more significant (Riahi et al., 2017). Given the limited current research on this topic, it seems important to investigate further the expected complex interactions between microplastics, nanoplastics, and climate response to properly assess their effective potential impact in the future.

### 7.3.3. Plastics as a stratigraphic marker

Microplastics have also gained significant attention as potential markers of the Anthropocene era, characterized by profound human influence on Earth's geology and ecosystems (e.g., Corcoran et al., 2014; Ivar do Sul and Labrenz, 2020; Weber and Lechthaler, 2021). They have even been referred to as “technofossils” due to their widespread distribution and persistence in the environment (Chen et al., 2022). Compared to other markers of the Anthropocene, such as concrete, lead/aluminum, and other metal elements (Waters, 2016), plastics and microplastics are more abundant and extensively distributed (Wang et al., 2021b). As mentioned before, in the future sedimentary records, plastiglomerates and microplastic particles could be considered like an anthropogenic marker horizon of human pollution (e.g., Corcoran et al., 2014; Utami et al., 2023).

Commonly, for dating recent deposits, the  $^{210}\text{Pb}$  method is used at the centennial scale, often combined with  $^{137}\text{Cs}$  dating (e.g., Turner et al., 2019). In the future, assemblages of microplastics could serve as alternative signature for dating sediment cores (Chen et al., 2022) with quite precisely age ranges for post-1910s, post-1930s, post-1950s, and post-1970s (Chen et al., 2022). With the availability of new time constraints at centennial scale resolution, building high-resolution sedimentary budgets for recent deposits becomes more achievable. This information seems quite valuable for discussions on the impacts of climate change on erosion processes, among other environmental phenomena at higher than centennial scale. Moreover, microplastic dating can help downscaling laws from event to centennial signatures. It would give insights on the validity of the quaternary analogy generally used for understanding past geological processes over geological time scales. The use of microplastics as markers thus opens an exciting opportunity to enhance our understanding of the Anthropocene and the geological implications at various temporal scales.

### 7.3.4. New constrains on physical processes

Studying the aggregation dynamics of plastics goes beyond microplastic research; it offers a gateway to unraveling the intricate physical

processes occurring in various fluid environments. This understanding is crucial for assessing the long-term fate and potential impacts of all particles (e.g., organic matter, mud) in geologic settings, providing essential information for environmental assessments and furthering our knowledge of fluid dynamics and sedimentary processes.

Research on plastic particle deformation and change of state during transport will also extend beyond microplastic-focused investigations. It will offer insights into the fragmentation of various sedimentary particles, ranging from pebbles to silt and mud, of from leaf to organic matter fragments. It will thus deepen our understanding of particle dynamics, sediment fragmentation, and transport processes along the land-to-sea continuum, but also to design solutions for plastic pollution remediation.

## 8. Conclusion

Through the eye of geoscientists, the present article aims to provide a comprehensive overview of the current state of knowledge concerning microplastics along the source-to-sink profile, shedding light on their specificities and interactions as particles in the environment and associated challenges.

The accuracy and reliability of microplastic data hinge on how well the sampling method captures the true microplastic distribution in the environment studied. Several issues have been listed on their quantification, this topic is essential for future monitoring, and to be able to close budgets and follow their evolution in space and over time. Indeed, the definition of microplastic subclasses (e.g., VSMP) is crucial to improve our understanding of transport dynamics and to effectively integrate the sediment-plastic budget.

The provided overview of microplastic sources, sinks, pathways and concentration along the source-to-sink (S2S) profile also highlights how by combining quantitative stratigraphy and S2S approaches with the study of plastics pollution, there is a potential for significant progress in understanding and managing plastic pollution's impact on the environment and human society.

Source-to-sink approaches can play a crucial role in enhancing our understanding of the plastic cycle in the environment. As mentioned earlier, such approaches can provide valuable insights into the transfer coefficient or sedimentation rate estimates, which are essential for assessing the transport and fate of plastic particles within natural systems. Additionally, employing source-to-sink approaches allows for a comprehensive and holistic understanding of the plastic cycle, aiding in the improvement of waste management strategies, and the coordination of effective solutions to tackle plastic pollution and its environmental impacts.

Several key points currently emerged regarding the value of studying plastics to make insights for source-to-sink studies and stratigraphy, with (i) plastics as tracers, (ii) impact of plastics on climate change (iii) plastic as a marker of the Anthropocene, and (iv) understanding fluid physics and aggregation.

This review could then help fostering collaboration and cross-disciplinary exchange between “plastic” and “geosciences” fields.

### Declaration of competing interest

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

### Data availability

Data will be made available on request.

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