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Review

Microplastics in freshwater sediment: A review on methods, occurrence, and sources

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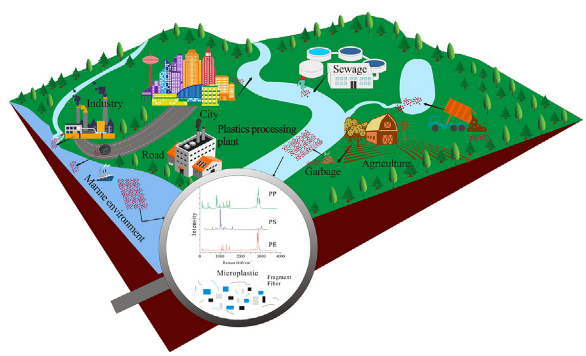
HIGHLIGHTS

- Microplastics occur ubiquitous in the investigated sediments of rivers, lakes and reservoirs at global scale.
- The abundance of microplastics in freshwater spans 2–5 orders of magnitude across different regions.
- Morphological characteristics indicate microplastics originate mainly from secondary sources.
- Morphological characteristics and chemical composition of freshwater and marine sediment microplastics are consistent.

GRAPHICAL ABSTRACT



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ABSTRACT

There is a rising concern regarding the accumulation of microplastics in the aquatic ecosystems. However, compared to the marine environment, the occurrence, transport, and diffusion of microplastics in freshwater sediment are still open questions. This paper summarizes and compares the methods used in previous studies and provides suggestions for sampling and analysis of microplastics in freshwater sediment. This paper also reviews the findings on microplastics in freshwater sediment, including abundance, morphological characteristics, polymer types, sources, and factors affecting the abundance of microplastics in freshwater sediment. The results show that microplastics are ubiquitous in the investigated sediment of rivers, lakes, and reservoirs, with an abundance of 2–5 orders of magnitude across different regions. Low microplastics concentration was observed in the Ciwalengke River with an average abundance of 30.3 ± 15.9 items/kg. In particular, an extremely high abundance of microplastics was recorded in the urban recipient in Norway reaching 12,000–200,000 items/kg. Fibers with particle size less than 1 mm are the dominant shape for microplastics in freshwater sediment. In addition, the most frequently recorded colors and types are white/transparent, and PE/PS, respectively. Finally, we conclude that the consistency of morphological characteristics and components of microplastics between the beach or marine sediments and freshwater sediments may be an indicator of these interlinkages and source-pathways. Microplastics in freshwater sediment need further research and exploration to identify its spatial and temporal variations and driving force through further field sampling and implementation of standard and uniform analytical methodologies.

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Contents

1. Introduction

The current period of human history is referred to as the Plastic Age (Cózar et al., 2014; Thompson et al., 2004). The material properties of plastics, especially their durability, lightness, and corrosion resistance, make them suitable for a wide range of applications (Lusher et al., 2017; Zeng, 2018; Zhang Y. et al., 2020). However, plastic debris has raised concerns due to its widespread distribution and associated environmental issues (Allen et al., 2019; Ding et al., 2017; Li J. et al., 2018). The plastics industry has developed rapidly since 1950, with global plastics output reaching 360 million tons in 2018 (Plastics Europe, 2019). The vast consumption and rapid disposal coupled with their physic-chemical properties such as very slow (bio)degradation rate as well as the inadequate and inappropriate collection and recycling of plastic waste are leading to a visible accumulation of plastic (Cózar et al., 2014; Strungaru et al., 2019). Large pieces of plastic debris can be partially removed from the environment and sent to the recycling process (Strungaru et al., 2019), while small plastic debris (<5 mm), usually defined as microplastics (de Souza Machado et al., 2018; Tagg et al., 2015), are not viable to remove from environmental matrices (Eerkes-Medrano et al., 2015). Therefore, for such small plastic particles, a more severe and widespread ecological risk is expected (Andrady, 2017; Silva et al., 2018).

Microplastics refer to plastic fibers, films, particles, etc. with a size of less than 5 mm, including primary microplastics and secondary microplastics (Hidalgo-Ruz et al., 2012; Thompson et al., 2004). Microplastics can be manufactured within this size range (primary microplastics) and can further be result of degradation and fragmentation of larger plastic items (secondary microplastics) (Ding et al., 2017; Fahrenfeld et al., 2019). Sewage treatment plants and runoff are considered important ways to transport land-based microplastics to freshwater and marine environments (Ziajahromi et al., 2017).

Microplastics have attracted public attention for their ubiquity and persistence in the aquatic environment, and the potential risk to the health of ecosystems (Cózar et al., 2014; Yukioka et al., 2020). Microplastics pollution have been found to reach some of the remote areas of the planet (Allen et al., 2019; Bergmann et al., 2019; Peeken et al., 2018; Jiang et al., 2019). Furthermore, due to its minute size, microplastics can be taken up by organisms from different trophic levels and with different feeding strategies (Karlsson et al., 2017; Lusher et al., 2017), thus, they can enter the food chain and accumulate at higher trophic levels (Ivleva et al., 2017). The negative effects of microplastics include physical damage to the gastrointestinal tract of ingesting organisms and toxicological effects caused by toxic chemicals and additives adsorbed in the microplastics, which can be carcinogenic, and endocrine-disrupted (de Souza Machado et al., 2018; Xu S. et al., 2020). Moreover, persistent organic pollutants, such as polycyclic aromatic hydrocarbons (PAH) and polychlorinated biphenyls (PCB) are also enriched on the surface of microplastics (Elert et al., 2017; Fahrenfeld et al., 2019; Van Cauwenberghe et al., 2015a; Zeng, 2018).

Sediments seem to be a long-term sink for microplastics (Li et al., 2020; Rochman, 2018). Logically, plastic with a density higher than

1.0 g/cm^3 sinks and gets deposits in the sediment, while low-density debris floats on the surface of the water or in the water column (Alam et al., 2019; Peng et al., 2018). Studies have suggested that the accumulation of biofilms, the adsorption, and accumulation of pollutants lead to an increase in the density of polymer debris, which is the main reason for the appearance of microplastics in sediments (Van Cauwenberghe et al., 2015a; Xu C. et al., 2020). High concentrations of microplastics have been recorded in beach sediment in Europe, North America, South America, Asia, and Africa (Alimba and Faggio, 2019; de Souza Machado et al., 2018; Yuan et al., 2019). However, to date, very few studies have focused on the presence of microplastics in freshwater sediment. Our understanding of microplastics pollution in freshwater sediments is relatively limited, with some basic issues such as the quantity, source, spatial distribution, and potential risk of microplastics remaining unresolved (Cózar et al., 2014).

Detailed studies have been carried out on freshwater sediments in a few rivers, lakes, catchments, and regions (Eerkes-Medrano et al., 2015; Li J. et al., 2018; Zhang et al., 2018). However, large sampling areas, long-term monitoring or a global perspective of microplastic pollution in freshwater sediment have not been implemented. As a result, we found that there is an urgent need to collate and compare current research findings and methodologies in microplastics of freshwater sediment to confirm the current state of knowledge and to sort out the pollution degree, characteristics, sources, and transmission of microplastics in freshwater sediments around the world. Thus, this paper focuses on five topics: (1) Provide an in-depth evaluation of current extraction and detection techniques used for microplastics in freshwater sediment; (2) Summarize the occurrence, characteristics of microplastics in freshwater sediments on a global scale; (3) Outline the potential sources, pathways of microplastics pollution in freshwater sediment; (4) Determine the factors affecting the distribution of microplastics in freshwater sediments; (5) Discuss effects and potential risks of microplastics in freshwater sediment.

2. Data collection

A search using the parameters "microplastic", "microbeads", "microfiber", "plastic debris", and "plastic fragment" in combination with "sediment" was performed on the ISI Web of Knowledge, Science Direct, and Google Scholar in February 2020. 38 works directly related to microplastics in freshwater sediments were selected, involving rivers, lakes, and reservoirs.

3. Microplastic analysis methodology

Although research on microplastics has been conducted for nearly two decades, the methodologies for sample collection, pretreatment, quantification, and identification have not been standardized (Li J. et al., 2018; Prata et al., 2019; Silva et al., 2018). The findings from different recorded studies vary widely, which makes it difficult to compare (Van Cauwenberghe et al., 2015a). These inconsistencies may be related to (i) the size and representativeness of the original sample selection, (ii) the sensitivity of the extraction technology selected, and (iii) inconsistent reporting units due to differences in sampling techniques (Ivleva et al., 2017). The overview of collection and analysis methods of microplastics in freshwater sediment is provided in Table 1 and Fig. 1.

3.1. Sample collection

The sampling methods of freshwater sediment can be divided into selective sampling, bulk sampling, and volume-reduced sampling, described in the methodologies published by Hidalgo-Ruz et al. (2012). In selective sampling, experimenters use tweezers to directly select microplastics that are identifiable by the naked eye from the field sediments. Generally, selective sampling is reasonable when the sample contains a relatively high amount of large microplastics (1–5 mm diameter), and is clearly distinguishable from the matrix (Ivleva et al., 2017). Selective sampling was applied in only one of the 38 freshwater sediment studies, which only targeted on microplastic pellets (Corcoran et al., 2015). Volume-reduced sampling refers to the method of reducing the volume of the bulk samples during the sampling process, leaving only the part of the sample that requires further processing (Hidalgo-Ruz et al., 2012). In bulk sampling, a certain weight or volume of sediment is collected in the field, not just microplastics. The benefits of bulk sampling are less time consuming, methodology standardization, and no special requirements for the sampling personnel (Alimba and Faggio, 2019; Zhang L. et al., 2020).

Sampling tools were recorded in 37 of the 38 reviewed freshwater sediment studies. Riverbank sediments/lakeshore sediment samples were taken with a stainless shovel, spoon, and spatula (Abidli et al., 2017) to collect bulk sediment samples from the surface or different depths of the riparian zone or lakeshore to determine the total abundance and vertical distribution of microplastics in the sediment (Jiang et al., 2019; Wen et al., 2018; Xiong et al., 2018). Collection of sediment from the below-water requires a vessel and a grab sampler that is lowered to the bottom of the river or lake to collect the samples (e.g. Peterson sampler, Van Veen grab) (Alam et al., 2019; Rodrigues et al., 2018).

The sampling unit is directly related to the sampling tool used (Hidalgo-Ruz et al., 2012). Approximately half of the studies use the area as the sampling unit, and sampling area varies from 250 cm^2, 400 cm^2 to 930 cm^2. Other sampling units are weight (from 0.2 to 5 kg) and volume of sediment (from 1 to 3.5 L). Ten studies do not specify the sampling unit. Considering that the weight of sediments varies greatly depending on the water content and type of sediments, MSDF recommends standardized sampling by volume (Directive, 2013). However, NOAA uses the weight of microplastics in the dried sediment to quantify the concentration of microplastics, so the quality can also be sampled by mass standardization.

The concentration of microplastics largely depends on the sampling location and depths and the distance from the center of human activity (Qiu et al., 2016). Besides, Besley et al. (2017) found that the concentration of microplastics collected from different depths varies greatly (Besley et al., 2017). The average microplastics concentration of 50 g samples collected from top 1–5 cm is higher than that of 50 g samples collected from top 2 and 10 cm (Besley et al., 2017). Due to its uneven distribution, an accurate estimation of microplastic concentration in sediment samples may require a definition of sampling depth (Besley et al., 2017; Prata et al., 2019). The sampling depths of different studies varied widely: in most of the studies, samples were collected from the top 2–3 cm or 5 cm of the sediment, while some studies sampled the top 10 cm or deeper profile of the sediment (Table 1). Besides, some studies did not define sampling depth in their study. The MSFD Guidance recommends sediment to be collected from the top 5 cm, with a minimum of 5 replicates, at least 5 m apart (Directive, 2013).

In the standardization protocol for sediment monitoring proposed by Frias et al. (2018), it is recommended that the location of the sampling sites should be 100 m parallel to the waterline (Frias et al., 2018; Stock et al., 2019). In freshwater environments, however, the range of banks or lakeshore is much smaller than that of the ocean, so sampling sites may be much closer to rivers and lakes. The sampling unit is recommended to be a 30 × 30 cm square with a sampling depth of 5 cm, collected with a metal shovel, and stored in a glass jar (Frias et al., 2018; Stock et al., 2019). For subtidal sediments, sampling by van Veen grabber or box corer is recommended (Stock et al., 2019). In order to prevent the collected sediment from being disturbed and to avoid loss of sediment, a drill corer or a Pürckhauer is recommended (Frias et al., 2018; Stock et al., 2019).

3.2. Sample preparation

In order to compare the actual concentration of microplastics in sediment without being affected by sample humidity, the samples should

Table 1
A summary of collection, pretreatment and analysis methods of microplastics in freshwater sediment.

Study area	Sample types	Collection	Depth and area or mass	Size (μm)	Density separator	Reference
Ciwalengke River, Indonesia	River	Ekman grab sampler	N/A	50um	NaCl	Alam et al., 2019
St. Lawrence River, Canada	River	Peterson grab	0–10 cm, 225 cm^2, 930 cm^2	500	Seving	Castañeda et al., 2014
Ottawa River, Canada	River	Ekman grab sampler	N/A, 3.5 L	100	NaCl	Vermaire et al., 2017
Wei River, China	River	Grab (B-10104, Ravenep)	N/A, 5 kg	75	NaCl	Ding et al., 2019
Nakdong River, South Korea	River	Stainless steel spoon	2 cm, 945 g	20	Lithium metatungstate	Eo et al., 2019
Brisbane River, Australia	River	Ponar stainless-steel grab sampler	3 cm	50	$ZnCl_2$	He et al., 2020
Rivers in the Tibetan Plateau, China	River	Stainless steel shovel	2 cm, 0.04 m^2, 200 g	45	$ZnCl_2$	Jiang et al., 2019
Rhine-Main River, Germany	River	Stainless steel spoon	3 cm, 30 cm^2	63	NaCl	Klein et al., 2015
Pearl River, China	River	Van Veen grab	5 cm, 2 kg	20	NaCl	Lin et al., 2018
Rhine River, Germany	River	Steel spade	7 cm, 0.01m^2	11	$ZnCl_2$	Mani et al., 2019
Bloukrans River, South Africa	River	N/A	5 cm, 2 kg	63	NaCl	Nel et al., 2018
Antuã River, Portugal	River	Van Veen grab	12 cm, 0.051m^2	N/A	$ZnCl_2$	Rodrigues et al., 2018
Beijiang River, China	River	Stainless-steel shovel	2 cm, 20 × 20 cm	47	NaCl	Wang L. et al., 2017
Wen-Rui Tang River, China	River	Peterson grab sampler	15 cm, 32 × 20 cm	20	$ZnCl_2$	Wang et al., 2018
Urban water in Changsha, China	River	Stainless-steel shovel	5 cm, 1000 g	50	$ZnCl_2$	Wen et al., 2018
Changjiang Estuary, China	River	Stainless steel spoons	5–10 cm	50	NaCl	Peng et al., 2017
Shanghai, China	River	Shovel	5 cm, 0.25m^2, 500 g	40	NaCl	Peng et al., 2018
Thames River, UK	River	Stainless steel scoop	10 cm, 1 L	N/A	$ZnCl_2$	Horton et al., 2017
Qin River, China	River	Grab dredge	5 cm	25	NaCl	Zhang et al., 2020
Ebro River, Mediterranean	River	Van Veen grab	10 cm, 0.046 m^2	63	NaCl	Simon-Sánchez et al., 2019
Ganga River, India	River	Steel spoon	10–15 cm 2–3 Kg	63	$ZnCl_2$	Sarkar et al., 2019
18 streams in and around the city of Auckland, New Zealand	River	Scoop	5 cm	63	NaI	Dikareva and Simon, 2019
Carpathian basin, Hungary	Lake	Van-Veen grab	2–3 kg	N/A	NaCl	Bordos et al., 2019
Lake Ontario, Canada	Lake	Selective sampling	5 cm	500	Sodium polytungstate	Corcoran et al., 2015
Taihu Lake, China	Lake	Peterson sampler	2 kg	50	NaCl	Su et al., 2016
Urban lake in London, UK	Lake	Piston corer	5 cm	100-	Sodium polytungstate	Turner et al., 2019
Dongting Lake, China	Lake	Stainless shovel	2 cm, 0.3 m × 0.2 m	50	$ZnCl_2$	Jiang et al., 2018
Qinghai Lake, China	Lake	Stainless-steel shovel	2 cm, 20 × 20 cm	100	Potassium formate	Xiong et al., 2018
Poyang Lake, China	Lake	Van Veen grab	0.25 m^2, 500 g	50	NaCl	Yuan et al., 2019
Lakes in Tibet plateau, China	Lake	Stainless-steel shovel	2 cm, 20 × 20 cm	100	Potassium formate	Zhang et al., 2016
Lake Bolsena, Italy	Lake	Stainless steel shovel	3 cm, 0.25m^2	300	NaCl	Fischer et al., 2016
Lake Chiusi, Italy	Lake	Stainless steel shovel	3 cm, 0.25m^2	300	NaCl	Fischer et al., 2016
Vesijärvi lake and Pikku Vesijärvi pond, Finland	Lake	Ekman sampler	5 cm	NO	NaCl	Scopetani et al., 2019
Nakdong River, South Korea	Lake	HTH gravity corer	10 cm	500	H_2O_2	Vaughan et al., 2017
Lagoon-Channel, Tunisia	Lake	Stain-less steel spatula	3 cm	50	Nacl	Abidli et al., 2017
Urban recipient in Norway	Lake	Van Veen grab	1 cm, 1 Kg	11	$ZnCl_2$	Haave et al., 2019
Three Gorges Reservoir, China	Reservoir	Van Veen grab	1 L, 0.25 m^2	45	NaI + NaCl	Di and Wang, 2018
Xiangxi Bay of Three Gorges Reservoir, China	Reservoir	Petersen grab sampler	0.0625 m^2	300	Potassium formate	Zhang et al., 2017

be dried to constant weight before analysis. The method to make sample dry can be divided into two types/patterns, including drying at high temperature with the temperature of 50 °C–100 °C and drying at room temperature with the temperature of approximately 25 °C. However, microplastics will deform and may break after been heated at high temperature (Horton et al., 2017; Qiu et al., 2016; Zobkov and Esiukova, 2017). According to the heat distortion temperature of the plastic, samples oven-dried at less than 60 °C can be a good choice, which is time-saving and can retain the physical form of microplastics in the sample (Nuelle et al., 2014; Zobkov and Esiukova, 2017).

3.2.1. Sample purification

Environmental samples contain a variety of organic matter (Prata et al., 2019). Purification procedures can be carried out directly on the environmental matrices to remove organic, inorganic particles, and debris, which can be mistaken for microplastics, leading to overestimation during quantification. Purification procedures are also necessary when single microplastics are analyzed (e.g. by FT-IR and Raman spectroscopy) for their polymeric matrix composition.

Purification process of samples can be divided into two main strategies, enzymatic degradation, and chemical degradation (Eo et al., 2019; Yuan et al., 2019; Zhang L. et al., 2020). In the chemical degradation method, microplastics samples are disposed with different chemicals, mainly 10% -30% hydrogen peroxide (H_2O_2) solution (Jiang et al., 2019; Li et al., 2020; Peng et al., 2017) or peroxide mixed with sulfuric acid (H_2SO_4).

Studies on the removal efficiency of organic matter by different acid, alkali, and enzyme digestion methods have shown that non-oxidizing acids (such as hydrochloric acid) have low efficiency in removing organic matter at room temperature with large amount of organic matter still remaining after digestion (Cole et al., 2015; Nuelle et al., 2014). Furthermore, sulfuric acid and nitric acid can destroy or damage the morphology of microplastics (Andrady, 2017; Besley et al., 2017; Li J. et al., 2018).

For enzymatic degradation, the microplastics samples are disposed with a mixture of technical enzymes (chitinase, proteinase, lipase, amylase, and cellulase) (Cole et al., 2015; Eerkes-Medrano et al., 2015). Organic matters such as carbohydrates, proteins, and lipids can be removed. In seawater samples rich in plankton, more than 97% (by weight) of organic matter can be removed within a few hours, while microplastics items were not affected (Li J. et al., 2018; Shruti et al., 2019). Since the storage and reaction temperature of the enzyme need to be strictly controlled and the price of the enzyme is expensive, the application of this method for large-scale routine sample purification still needs to be verified and improved.

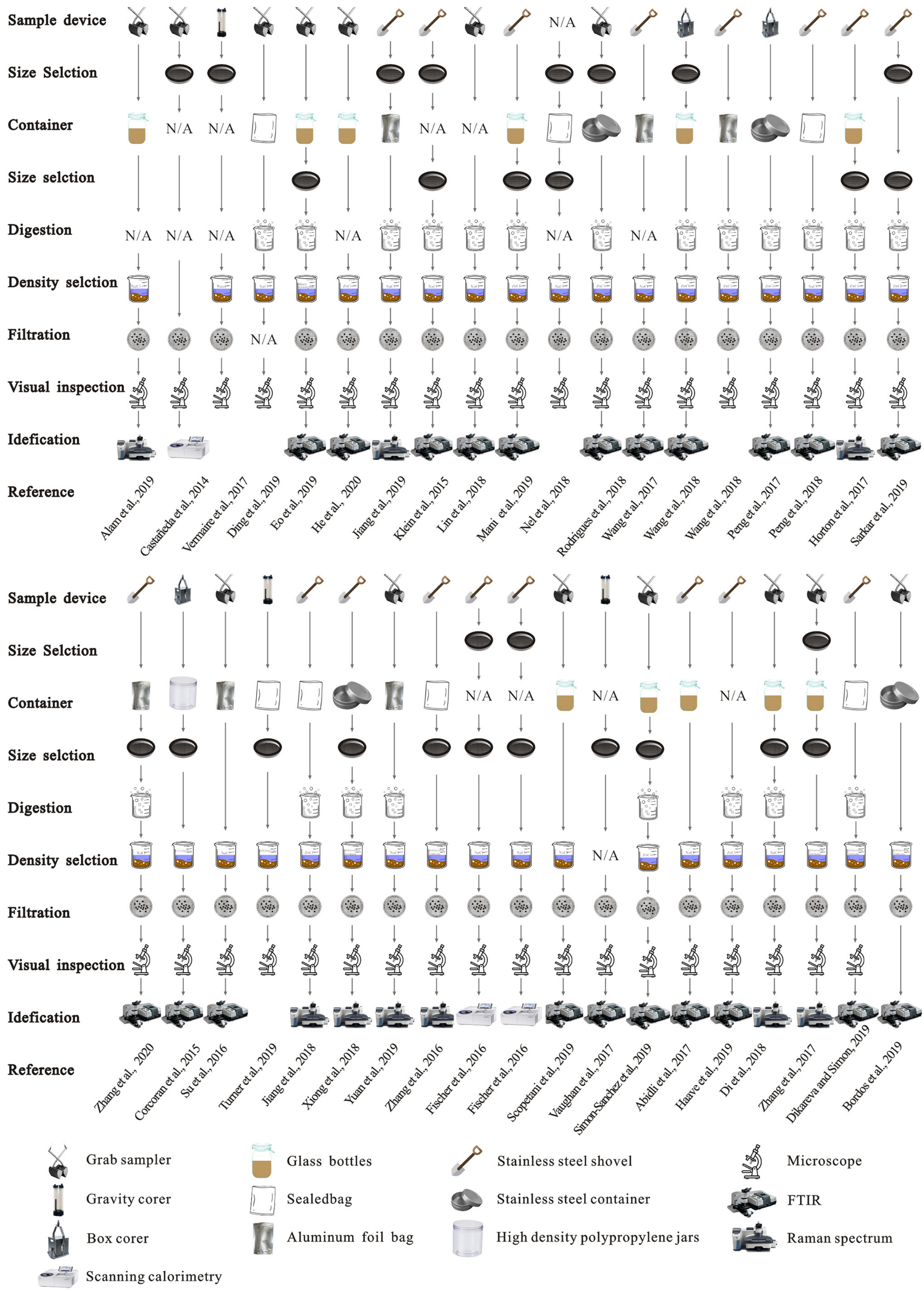

Fig. 1. Sampling, pretreatment and analysis methods used in the microplastic studies in freshwater sediment.

Consensus on effective methods is gradually being formed in the removal of organic matter, which is in the form of H_2O_2 digestion at a controlled temperature over a period of time (based on the quantity of organic matters) (Dümichen et al., 2015; Fahrenfeld et al., 2019; Zhang Y. et al., 2020). The study of Zobkov and Esiukova (2017) shows that the Fenton reaction is an efficient method for removing organic matter. Fenton reaction is an advanced oxidation process using H_2O_2 in the presence of a catalyst (Fe^{2+}), which can effectively degrade organic components that are usually difficult to degrade in H_2O_2 alone (Hurley et al., 2018). This demonstrates its superiority in removing all organic matter from complex environmental matrices (Hurley et al., 2018). In addition, after peroxide treatment, the size of most microplastic fragments remains unchanged, and the infrared spectroscopic identification of microplastics will not be affected (Tagg et al., 2015). It is noted that the temperature setting (70 °C) used in Fenton's reagent just exceeds the threshold tolerance of PA-6,6 polymer, which may cause oxidative damage and plastic polymer structure degradation (Hurley et al., 2018).

3.2.2. Sample separation

Microplastic analysis mainly includes three steps: (1) extraction (2) purification and (3) quantification and identification (Besseling et al., 2017; Dümichen et al., 2015). Sediment samples usually contain many interfering substances that can affect the quantification and identification of microplastics (Hale et al., 2020; Li J. et al., 2018). For bulk sediment samples, differences in density can be utilized to separate microplastics (0.80–1.45 g/cm^3) from the sediment matrix (2.65 g/cm^3) (Ivleva et al., 2017; Prata et al., 2019). This method mixes the sample with a high-density salt solution in the separation device and makes low-density particles such as the microplastics to float to the upper layer of solution, and finally separates microplastics (Li J. et al., 2018; Stock et al., 2019).

The saturated NaCl solution (density of 1.2 g/cm^3) is the most commonly used solution for the separating process because it is easily available, inexpensive, and non-toxic (Hidalgo-Ruz et al., 2012; Ivleva et al., 2017). However, higher density microplastics containing polyvinyl chloride (1.16–1.58 g/cm^3), polyformaldehyde (1.41–1.61 g/cm^3), and polyethylene terephthalate (1.38–1.43 g/cm^3) cannot be separated by sodium chloride solution. However, NaCl solution is still recommended by both the MSFD technical subgroup (2013), which excludes many higher density polymers from being analyzed (Li J. et al., 2018; Ivleva et al., 2017; Zobkov and Esiukova, 2017).

As a high-density (1.4 g/cm^3), non-toxic salt, sodium metatungstate (SPT) is also used in density separation of microplastics (Conkle et al., 2018; Horton et al., 2017; Li J. et al., 2018). However, even with this method, not all plastic can be separated, such as polyvinyl chloride (PVC) and polyformaldehyde (POM) (Corcoran et al., 2015; Eo et al., 2019; Turner et al., 2019). Moreover, the saturated polytungstate solution is quite expensive compared to other salts solutions.

Calcium chloride ($CaCl_2$, the density of 1.3 g/cm^3) and sodium iodide (NaI, the density of 1.8 g/cm^3) are also used for the separation solution of microplastics. The advantage of $CaCl_2$ is that it is cheap and non-toxic to the environment or humans. The study of Scheurer and Bigalke (2018) showed that $CaCl_2$ is not suitable for the separation of organic-rich samples (Scheurer and Bigalke, 2018). Since Ca^{2+} can bridge the negative charge of organic molecules, organic matter flocculates. Thus, the filter may be covered in thick brownish material, which interferes with the measurement (Scheurer and Bigalke, 2018). NaI solution gives good recoveries based on particle number, but which depends heavily on the type of plastic (Ivleva et al., 2017; Masura et al., 2015; Nuelle et al., 2014). When saturated sodium iodide (1.6 g/cm^3) is used as the density separation solution in elutriation devices, the recovery rate of microplastics can reach 65.8% -100% (Claessens et al., 2013). However, it is very expensive and needs to be cautious in handling.

Zinc chloride ($ZnCl_2$) solution (density of 1.6 g/cm^3) is widely used for the separation of microplastics, which can separate most types and different particle sizes of microplastics (Van Cauwenberghe et al., 2015a; Ivleva et al., 2017). By employing chloride solution and Munich Plastic Sediment Separator (MPSS), the recovery rate can reach 96–100% (by number) for larger microplastic (1–5 mm), and 96% (by weight) for small microplastic (<1 mm), respectively (Imhof et al., 2012). However, it is more environmentally hazardous compared to other salt solutions reported (Li J. et al., 2018; Zhang et al., 2018). Therefore, to minimize environmental pollution subsequent recovery and reuse of $ZnCl_2$ is necessary (Prata et al., 2019).

Potassium formate solution (1.58 g/cm^3) can also be used for the density separation solution of microplastics (Stock et al., 2019). It can separate various types of microplastics. Potassium formate is non-toxic, non-corrosive, and less harmful to the environment than $ZnCl_2$, while the cost of potassium formate is higher than that of $ZnCl_2$.

In addition to density-based methods, based on the lipophilic properties of plastic polymers also provides an alternative to separate microplastics (Crichton et al., 2017). Crichton et al. (2017) used canola oil and filtered water to obtain an average of 96.1% ± 7.4 recovery for both fibers and particles, with high recovery rates (94.9% ± 7.27) for high-density microplastics, such as polyvinyl chloride (PVC, density 1.16–1.58 g/cm^3) (Crichton et al., 2017). Olive oil has also been used in the extraction of microplastics in soil and compost samples. The recovery rate is also high with an average of above 90% from low polymers to high-density polymers (Scopetani et al., 2020a). Karlsson et al. (2017) increased the recovery rate by adding a drop of olive oil to the sodium chloride solution, which improved recovery rates from 64% to 82% (Karlsson et al., 2017). Oil extraction is cost-effective and environmentally friendly compared to other salt solutions (Crichton et al., 2017). It is noted that oil extraction requires a cleaning step with detergent.

Recovering microplastics larger than 1 mm is relatively easy, but current methods have certain limitations for assaying microplastics smaller than 1 mm (Fuller and Gautam, 2016). Only by stirring extraction, even if zinc chloride is used, the recovery rate of small particle size microplastics (<1 mm) is around 40% (Ivleva et al., 2017; Li J. et al., 2018). Some adjustments and strategies were proposed. The combination of separation of NaCl and NaI (Nuelle et al., 2014) can achieve good recovery of microplastics. The innovation of density separation instruments is also improving the capabilities of researchers to separate low-density microplastics and microplastics <500 μm (Eerkes-Medrano et al., 2015). Munich Plastic Sediment Separator (MPSS) established by Imhof et al. (2012) gives good recovery rate of large microplastic (5-1 mm) and small microplastic (<1 mm). The device can separate large-volume samples and up to 6 L of samples can be analyzed in one run (Imhof et al., 2012).

Another large-volume microplastic sample separation method, recommended by Felsing et al. (2018), Korona-Walzen-Scheider (KWS) utilizes the non-conductive properties of plastics to separate microplastics from the matrix and other particulate matter (Felsing et al., 2018). The device can separate samples with a volume of 20 cm × 15 cm × 20 cm at one run. This technique has good separation efficiency for microplastics of different sizes, densities, shapes, and ages (Felsing et al., 2018). It should be noted that this technique is for completely dry and unconsolidated samples. KWS shows very strong superiority for the separation of small microplastics, even for 63 μm microplastics, the recovery rate can reach 99%. However, this equipment only removes a part of the interferent, and still needs to remove the organic matter on the surface of the microplastics.

Optimizing Pressurized fluid extraction (PFE) can also be applied to extract microplastics. PFE is a widely used technique for the extraction of organic pollutants from multiple environmental matrices (Fuller and Gautam, 2016). At optimizing PFE conditions, microplastics would be either partially emulsifying or solubilizing, then microplastics can be extracted from environmental samples (Li J. et al., 2018). This separation technique is not affected by the particle size of the microplastics, and theoretically, even submicron particles can be analyzed (Fuller and Gautam, 2016). The morphology of microplastics and size distribution would be obliterated, which seems to be a limitation of this technique.

3.3. Analytical measurements

The suspected microplastics particles after density separation and purification need to be identified and quantified (Hidalgo-Ruz et al., 2012; Li J. et al., 2018). The most common strategy for detecting microplastics is to first identify the obvious/probable microplastics with the naked eye and then confirm by chemical composition analysis, which usually combined with optical and spectroscopic techniques (Besley et al., 2017; Silva et al., 2018) to avoid the occurrence of false negatives or minimize false positives (Prata et al., 2019).

Among the reviewed studies on freshwater sediment, a subset of the microplastics samples was instrumentally identified in 38 studies using Fourier-transform infrared spectroscopy (FTIR, 21 studies), Raman spectroscopy (10 studies), visual inspection (6 studies), or Pyrolysis gas chromatography mass spectrometry (pyro-GC/MS, 1 studies).

3.3.1. Visual inspection

Visual inspection of suspected microplastics particles as plastic is mainly based on physical characteristics such as shape and color (Fahrenfeld et al., 2019; Zhang L. et al., 2020). In all reviewed studies, visual examination is an essential step (Hidalgo-Ruz et al., 2012). The abundance and visual examination of microplastics are usually conducted under microscopes (Ivleva et al., 2017; Xu S. et al., 2020). The microscope used includes optical microscope (binocular biological microscope, dissecting microscope, and fluorescent microscope) and electron microscope (scanning electron microscope) (Hartmann et al., 2019; Karlsson et al., 2017; Qiu et al., 2016).

It is a good choice to observe microplastics directly on the filter paper without loss of microplastics due to transfer. So far, the stereomicroscope is the most forthright and thus widely used in manual counting and identification of microplastics. Several standardized criteria for strict and conservative inspection of microplastics are used (Hidalgo-Ruz et al., 2012; Norén, 2007; Zhang L. et al., 2020; Qiu et al., 2016; Peng et al., 2017; Prata et al., 2019). The key points are as follows:

(1). Small size (the largest dimension is less than 5 mm).
(2). In terms of shape, microplastics particles are roughly divided into five categories: fiber, pellet, foam, film, and fragment (Hidalgo-Ruz et al., 2012). The entire microplastic should show a uniform homogeneous thickness.
(3). The particle should have a relatively uniform color. If the particles are white or transparent, they should be examined under a microscope at high magnification.
(4). No cellular or organic structures are visible. If biofilm or other organic or inorganic is attached to the microplastic, it must be identified after removing the interference.

In many cases, the identification protocol also depends on personal visual determination or selection (Li et al., 2020; Qiu et al., 2016), which is open to bias and misjudgment. Moreover, visual counting suffers the defect of size limitation of the microscopy (Hidalgo-Ruz et al., 2012; Käppler et al., 2016; Zhang Y. et al., 2020). It is very difficult to visually detect microplastics that are less than 100 μm in size even using a microscope (Hanvey et al., 2017). The different sizes and shapes of microplastics may affect the results of their inspection and identification. Lenz et al. (2015) verified the visual inspection results of 452 fibrous and 857 fragment microplastics by instrument inspection. The results showed that among all the fragment microplastics, the accuracy rate of the particle with a size less than 50 μm was 63%, the accuracy rate of the part with a size between 50 and 100 μm was 67%, and the accuracy rate of the particle with a size greater than 100 mm was 83%. For the fibrous microplastics, the accuracy rate of fibers with a size of less than 1000 μm was 73%, the accuracy of fibers with a size between 1000 and 2000 μm was 76%, and the accuracy of fibers with a size greater than 2000 μm was 90% (Lenz et al., 2015). The results show that the number of false positives increases as the size of microplastics decreases. Besides, even for the most experienced experimenter, it can be difficult to distinguish between microplastics particles and other (in)organic matter like quartz particles, animal parts, or small plant pieces (Eerkes-Medrano et al., 2015; Li J. et al., 2018; Prata et al., 2019). Nonetheless, this strategy seems to be a reliable choice because of the difficulty transferring of small particles using tweezers, and some samples are large amounts and costly (Ivleva et al., 2017; Qiu et al., 2016).

3.3.2. FTIR spectroscopy

FTIR and Raman spectroscopy are highly recommended to identify microplastics (Hidalgo-Ruz et al., 2012). During the test, microplastics particles are excited so that the specific vibration of the structure can be detected (de Souza Machado et al., 2018; Song et al., 2015; Zhang Y. et al., 2020). The nature of the material (i.e., plastic and non-plastic) can be identified based on the generated characteristic spectrum and spectrum range (Elert et al., 2017; Imhof et al., 2012). The polymer identification is obtained by comparing the known reference spectra with obtained spectra (Li J. et al., 2018; Law and Thompson, 2014; Mai et al., 2018).

FTIR spectroscopy is an efficient, simple, and non-destructive detection technique with a comprehensive polymer database (Jung et al., 2018; Li J. et al., 2018). FTIR mainly has two modes of operation, namely reflection and transmission modes (Horton et al., 2017; Nuelle et al., 2014). Attenuated Total Reflectance (ATR) is widely used in the identification of microplastics. Larger microplastic particles (>500 μm) can be analyzed by ATR-FTIR (Ivleva et al., 2017; Tagg et al., 2015). The advantages of ATR-FTIR are that it can provide a strong signal-to-noise ratio and has an abundance of literature spectras. In addition, Jung et al. (2018) confirmed the use of clean wipers and water to remove surface residues is of great benefit in improving the quality of the ATR FT-IR spectrum (Jung et al., 2018). However, microplastics samples must be dried before spectral analysis (Qiu et al., 2016). ATR-FT-IR also seems to have some disadvantages in identifying microplastics. It needs for contact and pressure samples that can disrupt fragile microplastics. Besides, the microplastics must be moved from filters to rigid supports, which have a possibility of missing or damaged samples. ATR-FTIR imaging requires a lot of time and effort to find microplastic particles suitable for analytical work (Silva et al., 2018).

For smaller particles, FTIR must be used in combination with an optical microscope, the so-called micro-FTIR (Imhof et al., 2012; Zhang L. et al., 2020). As single particle analysis is not doable, microplastics particles are usually collected on a filter (Qiu et al., 2016). Micro-FTIR can analyze plastics in reflection or transmission mode. The typical size of plastics that can be analyzed by transmission can reach 10 μm (Simon et al., 2018). While, only microplastics samples of a certain thickness can be evaluated in transmission mode (Masura et al., 2015). Thick samples can be analyzed in reflection mode. Simultaneously, irregularly shaped microplastics will generate unexplained spectra caused by refraction errors (Song et al., 2015; Strungaru et al., 2019). Therefore, only microplastics with a certain thickness and regular shape can be analyzed in reflection mode. Otherwise, the signal gets disturbed/distorted due to reflection errors caused by light scattering. These disadvantages are difficult to avoid (Li J. et al., 2018; Ivleva et al., 2017; Mai et al., 2018).

Recently, FTIR imaging has been improved by applying focal plane array (FPA)-based detection (Ivleva et al., 2017; Primpke et al., 2017; Simon et al., 2018). Infrared map obtained by a focal plane array (FPA)-based imaging detect microplastics by scanning the surface of filters held microplastics (Hidalgo-Ruz et al., 2012). FPA-FTIR has high spatial resolution, i.e. 5.5 μm in Reflection and 1 μm in ATR mode. FPA imaging allows detection and identification of plastics smaller than 20 μm (Mintenig et al., 2017), and 5–10 μm is a more acceptable limit. This technology is not sensitive to thickness and is not interfered with filter membranes and impurities, which makes it an ideal model for the identification of microplastics (Loder et al., 2015). Thus, FPA-FTIR makes detailed and impartially high-throughput analysis of the entire filter rather than subareas of a filter possible (Chen et al., 2020; Li J. et al., 2018; Loder et al., 2015; Primpke et al., 2017). However, FPA analysis has the limitation that the sample should be purified and then concentrated in the filter before analysis (Chen et al., 2020).

3.3.3. Raman spectroscopy

Raman spectroscopy has been widely used in microplastic studies in freshwater sediment. The advantage of Raman spectroscopy is the high spatial resolution (Lenz et al., 2015; Rodrigues et al., 2018). In Raman, confocal microscopy resolution can easily be 1 μm. It also showed that the resolution required for submicroplastic (<1 μm) particles is achievable, and in some cases down to 500 nm (Anger et al., 2018). Furthermore, Raman spectroscopy allows the analysis of wet samples. One of the biggest drawbacks of Raman spectroscopy is the interference of fluorescence from (micro)biological, organic (e.g. humic substances), and inorganic (e.g., clay minerals) contaminations, which hampers the identification of microplastics (Ivleva et al., 2017; Eerkes-Medrano et al., 2015). Therefore, the samples should undergo a purification step before Raman analysis. Additionally, the choice of appropriate acquisition parameters (laser wavelength, photobleaching, laser power, magnification of the objective lens, measurement time, confocal mode) is important to circumvent the problem of strong fluorescence background (Horton et al., 2017; Imhof et al., 2012).

3.3.4. Mass spectrometry analysis

The thermoanalytical methods such as pyrolysis-GC/MS and TGA-MS have also been used for microplastic analysis. So far, only one study of freshwater sediment has used pyrolysis mass spectrometry (e.g. Py/GC/MS) for analysis (Castañeda et al., 2014). Py/GC/MS can identify the type of chemical component and the concentration of plastic type (Dümichen et al., 2015). Due to its destructive nature, thermal analysis cannot confirm the number or shape of particles (Li J. et al., 2018; Zhang Y. et al., 2020). The advantage of thermal analysis is that it does not require any sample preparation (Elert et al., 2017; Dümichen et al., 2015). However, this method has limitations on the size of microplastics. Microplastics samples smaller than 500 μm are difficult to handle as very small samples cannot be placed in a test tube. Furthermore, this method is not fit for analyzing samples mixed with high impurity concentrations (Ivleva et al., 2017; Silva et al., 2018).

Recently, methods have been proposed which provide for the mass analysis of some classes of polymers based on the preventive depolymerization and quantitative analysis of the resulting monomers, precisely to overcome the dimensional limitations connected to the analytical techniques, and in particular to the spectroscopic ones, more widely used for the analysis of individual microplastics. Wang L. et al. (2017) applied an alkali assisted thermal hydrolysis to quantify microplastics in the landfill sludge, which give good recoveries of 87.2–97.1% for PC and PET plastics particles (Wang L. et al., 2017). This method by quantifying the concentrations of the depolymerized building block compounds to determine the polycarbonate (PC)and PET microplastics in environmental samples (Wang L. et al., 2017). Castelvetro et al. (2020) applied this method to quantify poly (ethylene terephthalate) micro- and nanoparticle in sandy sediment, and the recovery reached 98.2% (Castelvetro et al., 2020).

3.3.5. Quality assurance and quality control

Quality assurance and quality control are essential for the analysis of microplastics. To assess the recovery of the method used, artificial manufacture plastic particles can be added as internal standards to natural samples of known quality before density separation (Zobkov and Esiukova, 2017). Zobkov and Esiukova (2017) recommend using fluorescent particles as internal standards. The fluorescent properties and obvious artificial shapes of fluorescent particles make them stand out from microplastics in natural sediments. They also supply a possible way to trace the lost microplastics during the analysis process. 50–500 μm fluorescent microplastics can be purchased from plastic products company. Larger particle size particles can be made from commercial products in daily life.

Leakage from the filter during sieving/transfer and adhesion in transfer and filtration devices are the main ways for the loss of microplastics during the experiment, especially microplastics with small particle size (Zhao et al., 2017). In the reviewed literature, most studies use only one density separation method solution to extract microplastics. The combination of NaCl and NaI can effectively improve the recovery rate of the experiment (Nuelle et al., 2014). Additionally, due to their lightweight, microplastics can easily adhere to the filter and transfer containers. Ultrapure water should be used to repeatedly wash the filter and transfer containers to ensure that all samples are transferred during the processing and transferring process.

Due to the wide range of sources and the accessibility of environmental contamination, quality control is particularly important for the credible quantification of microplastics. In all reviewed studies, quality control is an obligatory step. During the sample collection, pretreatment and analysis steps, precautions should be taken to avoid contamination (Mani et al., 2015; Su et al., 2016). Before use, all equipment should be thoroughly rinsed with ultrapure water (Eo et al., 2019; Lin et al., 2018). Samples, sub-sample, and filters are covered with aluminum foil after use to minimize microplastic contamination when exposed to air (Alam et al., 2019; Jiang et al., 2019). Use glassware as much as possible instead of plasticware; any plasticware is specified in the method description (Hartmann et al., 2019; Zhang et al., 2018). The blank is measured during the experiment, and the environmental blank is subtracted from the result to correct for background contamination (de Souza Machado et al., 2018; Fahrenfeld et al., 2019). The average value of the repeated samples is calculated as the measured concentration of the sample. Shunning any synthetic clothing during at all stages of the working process (Scopetani et al., 2020b). In harsh weather conditions, warm technical synthetic fabrics are needed, and cotton jumpsuits over warm synthetic clothes were worn to avoid contamination (Scopetani et al., 2020a). Besides, the experimenters wore nitrile gloves throughout the sampling, treatment, and analysis of the sample experiments (Ding et al., 2019; Scopetani et al., 2020b; Wen et al., 2018).

To date, the research community lacks a harmonization or standardization of quantitative units and methodologies, which do not allow for an insightful comparison on the occurrence of microplastics from different research communities (He et al., 2020; Yu et al., 2020; Rodrigues et al., 2018). The detection limits of microplastics vary greatly. Here, the limits depend heavily on the sampling and identification methodologies (Ivleva et al., 2017). For sieved sediment samples, the lower limit depends on the cut-off size efficiency of the filters used and usually focuses on 500 μm, 100 μm, 63 m and 50 μm. Furthermore, different quantitative units are used to represent the abundance of microplastics. For sediment, the quantitative units usually use items of microplastics per sediment area or sediment weight, with units such as items/m^2 or items/kg (Prata et al., 2019; Strungaru et al., 2019). However, some units cannot be transformed between each other, for example, items/kg and items/m^2, as they are measured in different dimensions (Yu et al., 2020). Thus, it is urgent to simplify and unify experimental methods, strengthen the comparative experiments of different methodology, and uniform quantitative units to improve the accuracy, reliability, repeatability, and comparability of the data recorded (Ivleva et al., 2017; Yu et al., 2020; Zhang L. et al., 2020).

4. Current knowledge of microplastics in freshwater sediment

4.1. Occurrence and abundance

The spatial distribution of freshwater sediment microplastic studies are shown in Fig. 2, summarizing the new developments in this research focus. Microplastics in freshwater sediment have been reported on a global scale, and research fields involve rivers, lakes, and reservoirs (Fig. 2). The overview of the abundance of freshwater sediment microplastics is presented in Table 2. The average abundance of freshwater sediment microplastics varies widely among different study areas and matrices.

4.1.1. Riverbank sediments

As an important way to transport microplastics from inland to the ocean, rivers collect nearby microplastics from plastic manufacturing plants (Mani et al., 2015; Rochman, 2018), plastic garbage, and sewage discharge (Ziajahromi et al., 2017). It is estimated that China is a hotspot of plastic pollution and is considered to be the largest riverine source of plastic to the ocean (Jambeck et al., 2015; Xu C. et al., 2020). Thus, microplastic pollution in China has received a lot of attention. Peng et al. (2018) observed the occurrence and distribution of microplastics in river sediments in Shanghai urban districts. The average microplastic concentration was observed to be 802 ± 594 items/kg (Peng et al., 2018). This study confirmed that the abundance of microplastics in densely populated areas was higher than that in sparsely populated areas (Peng et al., 2018). Wen et al. (2018) recorded the abundance of microplastics ranged from 270.17 ± 48.23 items/kg to 866.59 ± 37.96 items/kg in the urban water sediments in Changsha. Apart from the Yangtze River catchment, microplastic pollution in riverbank sediments was also reported in the Pear River catchment. The abundance of microplastics in Beijiang River ranged from 178 ± 69 to 544 ± 107 items/kg (Wang J. et al., 2017). The abundance of microplastic at 10 sites along the lower reaches of Qin River catchment ranged from 0 to 97 items/kg (Zhang L. et al., 2020). In riverbank sediments of the Tibet Plateau, an average microplastic concentration of 90–130 items/kg was illustrated (Jiang et al., 2019).

The occurrence of microplastics has been recorded in freshwater sediments around the world and the mean values of microplastics abundance in river sediment varied significantly from almost none to several tens of thousands of items per kilogram. The average abundance of microplastics in riverbank sediment of Thames River in UK was 660 items/kg (Horton et al., 2017). A similar concentration of microplastics was found in Bloukrans River in South Africa (Nel et al., 2018). In the sediments of river Ganga at eastern India, mesoplastics (>5 mm) and microplastics (<5 mm) particles with varying degree of the mass fraction (11.48–63.79 ng/g sediments), numerical abundance (99.27–409.86 items/kg) and morphotypes (Sarkar et al., 2019). So far, the highest reported value in riverbank sediment was recorded in Rhine-Main River, Germany, the abundance of microplastics ranged from 260 ± 10 to 11,070 ± 600 items/kg (Mani et al., 2019).

4.1.2. River bottom sediment

The presence of microplastics has been recorded in river bottom sediment worldwide. For example, in the river bottom sediment of the Ciwalengke River in Indonesia, microplastic concentration was reported to be 30.3 ± 15.9 items/kg, which may be due to industrial washing processes and household laundry activities (Alam et al., 2019). In the river bottom sediment of the Ottawa River in Canada, the average microplastics concentration was 220 items/kg (Vermaire et al., 2017). In Brisbane River in Australia, the mass fraction of microplastics varied from 0.18 to 129.2 mg/kg, and the calculated number of microplastics was 10 to 520 items/kg (He et al., 2020). Microplastics were also detected in Antuã River in Portugal, and this study emphasized the importance of rivers as a potential transportation system for microplastics (Rodrigues et al., 2018). In the study of Nakdong River, the mean abundance of microplastics in river bottom sediment was 1970 ± 62 particles/kg and estimated that the carrying capacity of microplastics of Nakdong River was 5.4–11 trillion by number, which is 53.3–118 tons by weight in 2017 (Eo et al., 2019). River deltas are considered as hotspots of microplastic accumulation. In the microplastic study of Ebro River in Mediterranean, a mean abundance of 2052 ± 746 items/kg was found in estuarine benthic sediments (Simon-Sánchez et al., 2019). Traces of microplastics are found not only in rivers but also in small streams. Microplastics were found in 18 streams in and around the city of Auckland, and the abundance of microplastics are similar to that found in larger systems (80 items/kg in sediment) (Dikareva and Simon, 2019).

More recently, studies reported that high concentrations of microplastics have been found in river bottom sediment of China, for example, Changjiang Estuary (20–340 items/kg), Wei River (360–1320 items/kg), and Pearl River (80–9597 items/kg) (Ding et al., 2019; Lin et al., 2018; Peng et al., 2017). In particular, an extremely high concentration of microplastics (32,947 ± 15,342 items/kg) was detected in river bottom sediment of Wen-Rui Tang River in China (Wang et al., 2018). In St. Lawrence River, the mean density of microplastic was 13,832 ± 13,677 microbeads/m^2 (Castañeda et al., 2014).

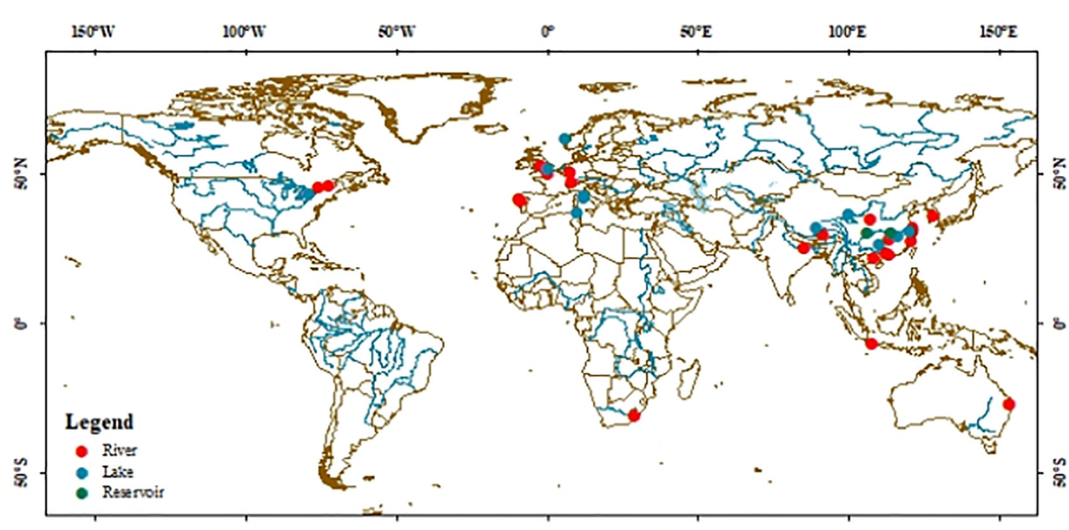

Fig. 2. Global studies of microplastics pollution in freshwater sediment published to February 2020 (based on 38 studies). For further details on microplastic abundance and characterization in these studies refer to Table 2.

Table 2
A summary on abundance and characteristics of microplastics in freshwater sediment.

Study area	Abundance	Shape	Size	Color	Chemical component	Reference
Ciwalengke River, Indonesia	30.3 ± 15.9 items/kg	91% fiber	50–100 μm (34%), 300–500 (18%); 500–1000(18%)	N/A	Polymer mixture (44%); PET (33%), PA (22%)	Alam et al., 2019
St. Lawrence River, Canada	13,832 ± 13,677 items/m^2	90% pellet	0.4 to 2.16 mm	Black:>90	PE	Castañeda et al., 2014
Ottawa River, Canada	220 items/kg	N/A	0.5 to 3 mm	N/A	N/A	Vermaire et al., 2017
Wei River, China	360–1320 items/kg	42% - 53% fiber	31.2% <0.5 mm	N/A	PE, PVC, PS	Ding et al., 2019
Nakdong River, South Korea	1970 ± 62 items/kg	>50% fragment	74% < 300 mm	N/A	PP (24.8%) + PE (24.5%)	Eo et al., 2019
Brisbane River, Australia	10–520 items/kg	Films dominant:	<3 mm	White dominant	PE: 70%	He et al., 2020
Rivers in the Tibetan Plateau, China	90–130 items/kg	53.8% - 80.6% fiber	>70% <1 mm	While:>50%	PET (50%), PA (17%), PE (12%), PS, PA	Jiang et al., 2019
Rhine-Main River, Germany	228–3763 items/kg	>50% pellet	630–5000 μm	N/A	PE: 50%	Klein et al., 2015
Pearl River, China	80–9597 items/kg	54.7% fiber	65.3% 0.02–1 mm	36% yellow, 26.8% white, 11.7% Black	PE (47.6%), PP (26.2%)	Lin et al., 2018
Rhine River, Germany	260 ± 10 to 11,070 ± 600 items/kg	N/A	96.3 ± 5.7% <75 μm	N/A	APV:70%	Mani et al., 2019
Bloukrans River, South Africa	160.1 ± 139.5 items/kg	N/A	N/A	N/A	NA	Nel et al., 2018
Antuã River, Portugal	2.6–629 items/kg	Fragments (dominant), pellets, films, foam	N/A	Colored dominant	PE (29.4%), PP (29.4%)	Rodrigues et al., 2018
Beijiang River, China	178 ± 69 to 544 ± 107 items/kg	N/A	N/A	N/A	PE, PP	Wang J. et al., 2017
Wen-Rui Tang River, China	32,947 ± 15,342 items/kg	65% fragments	<300 μm 68.3–94.8%, 20–100 μm 68%	N/A	N/A	Wang et al., 2018
Urban water in Changsha, China	270.17 ± 48.23–867 ± 38 items/kg	50.82% fragment:	70% < 1 mm	Transparent dominant (35%)	PS (29.4%), PET (17.4%)	Wen et al., 2018
Changjiang Estuary, China	20 to 340 items/kg	93% fiber	31.19% <100 μm; 62.15% 500–1000 μm	Transparent (42%)	Rayon, PES, AC, PET, PS	Peng et al., 2017
Shanghai, China	802 ± 59 items/kg	89% pellet	most of <1 mm	White (90%)	PP (57%) + PET (17%)	Peng et al., 2018
Thames River, UK	660 items/kg	49.3% fragment, 47.4% Fiber	1–4 mm	N/A	PET (41%), PP (15%), PE (15%), PS (6%), PVC (3%)	Horton et al., 2017
Qin River, China	0–97 items/kg	Fibers (30.9%), sheets (62.8%), fragments (6.3%)	1–5 mm (76.0%)	Black (1.5%), white (30%), blue (27.6%), green (18.3%), red (18.5%)	PP (55.3%), PET (21.3%), and PE (17.0%)	Zhang et al., 2020
Nakdong River, South Korea	250–300 items/kg	>50% film	N/A	N/A	N/A	Vaughan et al., 2017
Ebro River, Mediterranean	2052 ± 746 items/kg	>50% fiber	>1000 μm (33.6%),<1000 μm, 41.8%	Colored (54.0%), transparent (35.3%), Black (33.8%)	PA (24%), PE (16%), acrylic (12%), PET (12%), PP (8%)	Simon-Sánchez et al., 2019
Ganga River, India	99.27–409.86 items/kg	N/A	N/A	N/A	PET (39%), PE (30%), PP (19%)	Sarkar et al., 2019
18 streams in and around the city of Auckland, New Zealand	9–80 items items/kg	Fragment (39%), fiber (34%)	63–500 μm dominate	Black,	PP, PE, PET, PVC	Dikareva and Simon, 2019
Carpathian basin, Hungary	0.46–1.62 items/kg	N/A	N/A	N/A	PP (55%), PE (11%), PS (30%)	Bordos et al., 2019
Lake Ontario, Canada	Total 4635 items	>65% pellets	0.5–3 mm	>65% white	PE:74%	Corcoran et al., 2015
Taihu Lake, China	11–235 items/kg	48–84% fiber	>65% 100–1000 μm	White and transparent (29–44%)	Cellophane dominant	Su et al., 2016
Urban lake in London, UK	539 items/kg	>80% fiber	500-μm to 1 mm dominant	Blue 25%, white 22%, Red 17%	PS, PA	Turner et al., 2019
Dongting Lake, China	200–1566 items/kg	12–77.4% fiber	<0.5 μm	Transparent (30%) + blue (25%) + red (22%)	50% PET	Jiang et al., 2018
Qinghai Lake, China	67 to 1292 items/m^2	Fragment and fiber>85	100–500 μm >50%	transparent:75%	>50% PE	Xiong et al., 2018
Poyang Lake, China	54–506 items/kg	44% fiber	57.1% < 0.5 μm	Colored:36%	37% PP+ 30% PE	Yuan et al., 2019
Lakes in Tibet plateau, China	4–1219 items/m^2	>80% fragment	1–5 mm	Colored	PP + PE>90%	Zhang et al., 2016
Lake Bolsena, Italy	112 ± 32 items/kg	Fragment and fiber	<0.3 mm	N/A	N/A	Fischer et al., 2016
Lake Chiusi, Italy	234 ± 85 items/kg	Fragment and fiber	0.3 to 0.5 mm	N/A	N/A	Fischer et al., 2016

Table 2 (*continued*)

Study area	Abundance	Shape	Size	Color	Chemical component	Reference
Vesijärvi lake and Pikku Vesijärvi pond, Finland	395.8 ± 90.7 items/kg	NA	N/A	N/A	53% PA	Scopetani et al., 2019
Lagoon-Channel, Tunisia	704 ± 111 to 1483 ± 19 items/kg	Fiber (91%), fragments (5%), films (3%)	(0.1–1 mm) 56%- 97%	>50% Back	PP, PE	Abidli et al., 2017
Urban recipient in Norway	12,000–200,000 items/kg	N/A	<25 μm 56.3–70.1%	N/A	PA, PP, PS	Haave et al., 2019
Three Gorges Reservoir, China	25–300 items/kg	33.9–100% fibers	500–1000 μm 1.7% to 77.8%	>50% transparent	39% PS, PP 29%, 21% PE	Di and Wang, 2018
Xiangxi Bay of Three Gorges Reservoir, China	80 to 864 items/kg	70% fragment	1–5 mm 71.4%	55.6% Blue	80% PP	Zhang et al., 2017

4.1.3. Lakeshore sediment

Lakes are relatively closed, natural water ponds that can store water from precipitation, surface runoff, and groundwater. Plastic garbage generated in the lake catchment can be transported to the lake and accumulated there (Su et al., 2016; Zhang Y. et al., 2019). In 2018, a high concentration of microplastics was detected in the lakeshore sediment of Dongting Lake in China (Jiang et al., 2018). Abidli et al. (2017) provided unequivocal evidence of microplastics pollution in the sediment of Lagoon-Channel in Tunisia. In particular, an extremely high abundance of microplastics in lake shore sediment was recorded in urban recipient in Norway, and the abundance reached 12,000-200,000 items/kg (Haave et al., 2019). The major source of microplastics in the urban recipient in Norway was considered to be the sewage outlets (Haave et al., 2019). In the remote area of the Tibet Plateau, a high concentration of microplastics was found (4–1219 items/m^2), which mainly derived from mismanaged plastic wastes (Zhang et al., 2016; Xiong et al., 2018).

4.1.4. Lakes bottom sediment

The concentration of microplastics in lake sediments was concentrated in the range of 11–2175 items/kg. In 2015, a high abundance of microplastics was detected in lakes bottom sediments of Taihu Lake, China (Su et al., 2016). Turner et al. (2019) provided a temporary sediment record of microplastics in an urban lake of London, and the average concentration of microplastic was 539 items/kg (Turner et al., 2019). A similar microplastics concentration (54–506 items/kg) was found in Poyang Lake in China (Yuan et al., 2019). Recently, microplastics were found in the Central Eastern European inland lakes (0.46–1.62 items/kg), which was also the first results indicated that fish ponds can act as deposit areas for microplastics (Bordos et al., 2019). High abundance of microplastics were also found in sediments from Lake Bolsena, Italy (112 ± 32 items/kg), Lake Chiusi, Italy (234 ± 85 items/kg), Vesijärvi Lake, and Pikku Vesijärvi pond in Finland (395.8 ± 90.7 items/kg) (Fischer et al., 2016; Scopetani et al., 2019). A report showed that a total of 4635 microplastic debris were found in Lake Ontario in Canada (Vermaire et al., 2017).

4.1.5. Reservoirs sediments

Due to water storage, reservoirs could be potential areas for the accumulation of microplastic debris (Di and Wang, 2018; Watkins et al., 2019; Zhang et al., 2018). Located in an economically prosperous region, the Three Gorges Reservoir is the largest reservoir in China (Zhang et al., 2017). Two studies on sedimentary microplastics pollution were conducted in the Three Gorges Reservoir (TGR). Microplastics abundance in sediment varied from 25 to 300 items/kg in the Yangtze River mainstream and 80 to 864 items/m^2 in the tributary estuaries. The continuous efforts of sewage treatment and pollution prevention systems can hardly keep up with the development of cities and industries along the river, which seems to be the cause of pollution in the Three Gorges Reservoir (Di and Wang, 2018; Zhang et al., 2017).

4.2. Features of microplastics in freshwater sediments

Unlike traditional environmental pollutants, microplastics exist in the environment in different shapes, sizes, densities, colors, and polymer types, as well as other inherent and attached pollutants (Li et al., 2020; Zhang et al., 2018; Zhang Y. et al., 2020). The impact of microplastics on the environment and its fate are closely related to these characteristics (de Souza Machado et al., 2018). The deposition of microplastics in sediments is related to their size, density, and shape (Besseling et al., 2017). Hence, in addition to its abundance, characteristics of microplastics have also been recorded.

4.2.1. Shapes

Microplastics in the environment occur in a variety of shapes and sizes. Common shapes of microplastic include pellet/spherule, fragment/sheet, foam, fiber/line, and film (Akhbarizadeh et al., 2018; Hidalgo-Ruz et al., 2012). These shapes depend on the original form of primary microplastics, the erosion and degradation processes of the plastic particle surface, and residence time in the environment (de Souza Machado et al., 2018; Zhang Y. et al., 2020). Various shapes of microplastics including fragment, foam, fiber, and film have been detected in the freshwater sediments. The fiber is the most common shape of microplastic in freshwater sediments. In sediment collected from Ciwalengke River, fibrous microplastics were more dominant (91%), compared to fragment (9%) (Alam et al., 2019). In the sediments of Changjiang Estuary, fiber was the most prevalent shape (93%) among all microplastic particles (Peng et al., 2017). However, in Shanghai (urban centers) and St. Lawrence River, the dominant shape of microplastics detected in sediment was pellet, contributed to approximately 90% of the total particle numbers and only 5–8% comprised fibers (Castañeda et al., 2014; Peng et al., 2018). Sediment studies in Nakdong River, Wen-Rui Tang Rive, Lakes in Tibet plateau, and other two studies also suggested that the main form of microplastics was fragment (Eo et al., 2019; Wang et al., 2018; Zhang et al., 2016). However, the most common shape of microplastics in Brisbane River was film, followed by fragment and fiber (He et al., 2020).

Shape, to a large extent, can infer the initial material of microplastics, as certain shapes may be derived from specific products (Andrady, 2017; Zhang Y. et al., 2020). Fiber, for example, is the predominant shape detected in the sediment of the Changjiang Estuary, which is likely closely connected to textiles. Washing is an important pathway that releases them into the environment (Peng et al., 2017). While fragmented microplastics could possibly originate from the exposure of larger plastic items to strain, fatigue, or UV light (Wen et al., 2018; Xiong et al., 2018; Zhang et al., 2016). The surface texture of fragments and pellets (for example, grooves, cracks, adhered particles, and flakes) provides good evidence of mechanical wear and chemical weathering to produce microplastics (Su et al., 2016; Zhang et al., 2016). The film mainly comes from plastic bags and packaging materials. Foam-shaped microplastics come from the damage of styrofoam (Xu S. et al., 2020; Zhang et al., 2018). Pellet is likely to virgin pellets spilled during transportation and processing (Corcoran et al., 2015), or as spherule and microbeads used in cosmetic products and sandblasting media, and in air-blasting agents or in industrial cleaner (Fahrenfeld et al., 2019; Hartmann et al., 2019; Mani et al., 2015).

4.2.2. Size

The particle size of microplastics directly affects their migration in the water environment and whether they can be ingested by organisms, which is closely related to biosafety (Akhbarizadeh et al., 2018; Li et al., 2020; Zhang Y. et al., 2020). At this stage, the size range of microplastics varied greatly and the smallest size of detected microplastics becomes smaller with technological innovation (Besseling et al., 2017; Hartmann et al., 2019). Generally, microplastics with a particle size of less than 1 mm are more abundant in freshwater sediments and as particle size increased the microplastic abundances show a trend of decrease (Corcoran et al., 2015; Ding et al., 2017). For example, in Ciwalengke River, the majority of size of microplastics was 50–100 μm (34%), followed by 300–500 μm (18%) and 500–1000 μm (18%); while larger particles (1000–2000 μm) accounted for a smaller proportion (Alam et al., 2019). In Nakdong River, the particle size range of 100–150 μm was the largest proportion in sediment with the average and median values of 248 μm and 155 μm, respectively (Eo et al., 2019). However, in sediment samples of the in Xiangxi Bay of Three Gorges Reservoir, Thames River, and Qinghai Lake, the size of the microplastics was larger (1-5 mm), which may be related to its source (Xiong et al., 2018; Zhang et al., 2016; Zhang et al., 2017).

4.2.3. Color

As colored microplastics particles are easily mistaken for food by aquatic organisms (He et al., 2020; Rodrigues et al., 2018), color is one of the focus of microplastics research. Various colors of microplastics have been recorded including white, transparent, red, yellow, green, brown, gray, etc. (Cheung and Fok, 2017; Cole et al., 2015). Furthermore, colors are expected to identify potential sources of microplastics and potential contamination during sample preparation (Fahrenfeld et al., 2019; Zhang L. et al., 2020). Transparent microplastics are usually derived from disposable plastics, such as plastic bags, disposable plastic cups, and bottles, which are disposable and have short lifetimes (Li et al., 2020; Prata et al., 2019; Xiong et al., 2018). Colored microplastics are likely to originate from a variety of plastic consumer products with a long service life. (Andrady, 2017; Eo et al., 2019). As color is not permanent and bleaching processes can occur in the sample preparation process (Li et al., 2020; Yuan et al., 2019). Discussion of color to deduce the type or origin of microplastics must be cautious.

The discussion on color is not very unanimous at present. Half of the studies on freshwater sediment have no description of the color of microplastics. While divergence still exists in the rest of the studies discussed the color of microplastics. Some research communities discuss the color of microplastics directly based on the results recognized by the naked eye. However, it has been suggested to group microplastics into four obvious colors (transparent, black, white, and colored); instead of evaluating other more controversial colors (e.g., yellow, green, blue, etc.) (Jiang et al., 2019; Zhang et al., 2016).

The most common colors observed in microplastics of freshwater sediments were white and transparent. White microplastics made up the majority of microplastics in the sediments of Brisbane River (He et al., 2020). 90% of the microplastics detected in urban rivers sediment of Shanghai were white (Peng et al., 2017). Transparent was the dominant color in surface sediments of urban water in Changsha (Wen et al., 2018). Colored microplastics were also very common. For example, in Antuã River, the majority of the color group in sediment samples was colored one, followed by black, white and transparent (Rodrigues et al., 2018). In addition, multiple colors of microplastics were observed in Pearl River and particles were yellow (36.2%), white (26.8%), and black (11.7%) (Lin et al., 2018).

4.3. Polymer types

Chemical composition is the most basic criterion for defining microplastic pollution (Zhang Y. et al., 2020). Plastics are synthetic polymers made from a variety of compounds with different characteristics (Hidalgo-Ruz et al., 2012). The most common types of plastic polymers are polyethylene (PE), polypropylene (PP), polystyrene (PS), polyvinyl chloride (PVC), and polyester (PET) (Lusher et al., 2017).

Thirty-three of the studies record the chemical composition of microplastics, although not all of them implemented severe chemical analyses. For microplastics in freshwater sediment, PE is the main polymer followed by PP and PS, and the chemical composition varies greatly in different regions (Table 2). For example, in sediments of St. Lawrence River, the melting point of microplastic pellets is 113.7 °C, which suggests a polyethylene composition (Castañeda et al., 2014). In Ciwalengke River, polyethylene terephthalate (PET) and Polyamide (PA) dominate the type of particles observed and account for 33% and 22%, respectively (Alam et al., 2019). Synthetic compounds in Nakdong River include PP, PE, PES, PVC, PS, acrylic, polydimethylsiloxane, PU, poly(acrylate-styrene) and poly (lauryl acrylate) and their proportions are 24.8% 24.5%, 5.5%, 5.4%, 5.3%, 4.6%, 4.5%, 3.9%, 3.7%, and 3.6%, respectively (Eo et al., 2019). In Brisbane River, 70% of microplastics from sediments are identified as PE (He et al., 2020). In Changjiang Estuary, a total of six polymer types were identified including rayon (RY), acrylic (AC), polyester (PES), ethylene propylene diene monomer (EPDM), polyethylene terephthalate (PET), and PS (Peng et al., 2017). However, the majority of microplastics found in Taihu Lake were cellophane, followed by polyethylene terephthalate and polypropylene (Su et al., 2016). So far, there is no clear correlation or explanation for the variability of composition of the polymer types in freshwater sediment. Further research is needed to establish if there is a predominant group of polymers occurring in microplastics pollution of freshwater sediment and whether this polymer composition changes due to sample location and particle distance traveled (Eerkes-Medrano et al., 2015; Fahrenfeld et al., 2019; Hartmann et al., 2019).

4.4. Comparison with microplastic from marine sediment and freshwater

Just like in freshwater sediment, microplastics are common in marine sediment and surface water of freshwater systems on a global scale. From the above sections, we notice that the characteristics of microplastics in freshwater sediment vary greatly among rivers, lakes, and reservoirs. This difference also exists for microplastics in the freshwater environment and marine environment. A comparison with microplastics features in freshwater sediment, marine sediment, and surface water of freshwater systems is presented in the Fig. 3.

Examination of the size distribution of microplastics on marine sediments reveals that the particle size of microplastics mainly distributes in 1–5 mm (Van Cauwenberghe et al., 2015b). For example, along the beaches of the southeast coast of India, the largest proportion of microplastics is 2.36–4.75 mm both in the high tide line (87%) and low tide lines (86%) (Karthik et al., 2018). However, in offshore sediment of the Yellow Sea and the East China Sea, 89% of microplastic are less than 1000 μm (Zhang C. et al., 2019). Isobe et al. (2015) found that as the size of microplastics increases, the abundance of microplastics decreases (Isobe et al., 2015). However, when the size of the microplastic is less than 1 mm, as the size of the microplastic decreases, its concentration decreases rapidly (Isobe et al., 2015). Moreover, the size of microplastics reported in individual studies has been determined by the size range of the microplastic sampled and identification methods (Zeng, 2018; Zhang Y. et al., 2020). For example, in the freshwater environment, the average microplastic size of samples collected using a sieve or trawl with a pore size of 200–1000 μm is 1 mm to a few millimeters; however, samples collected by using a smaller mesh size (50–63 μm), the average size of microplastics seem to be with an average size of less than 700 μm (Zhang Y. et al., 2020).

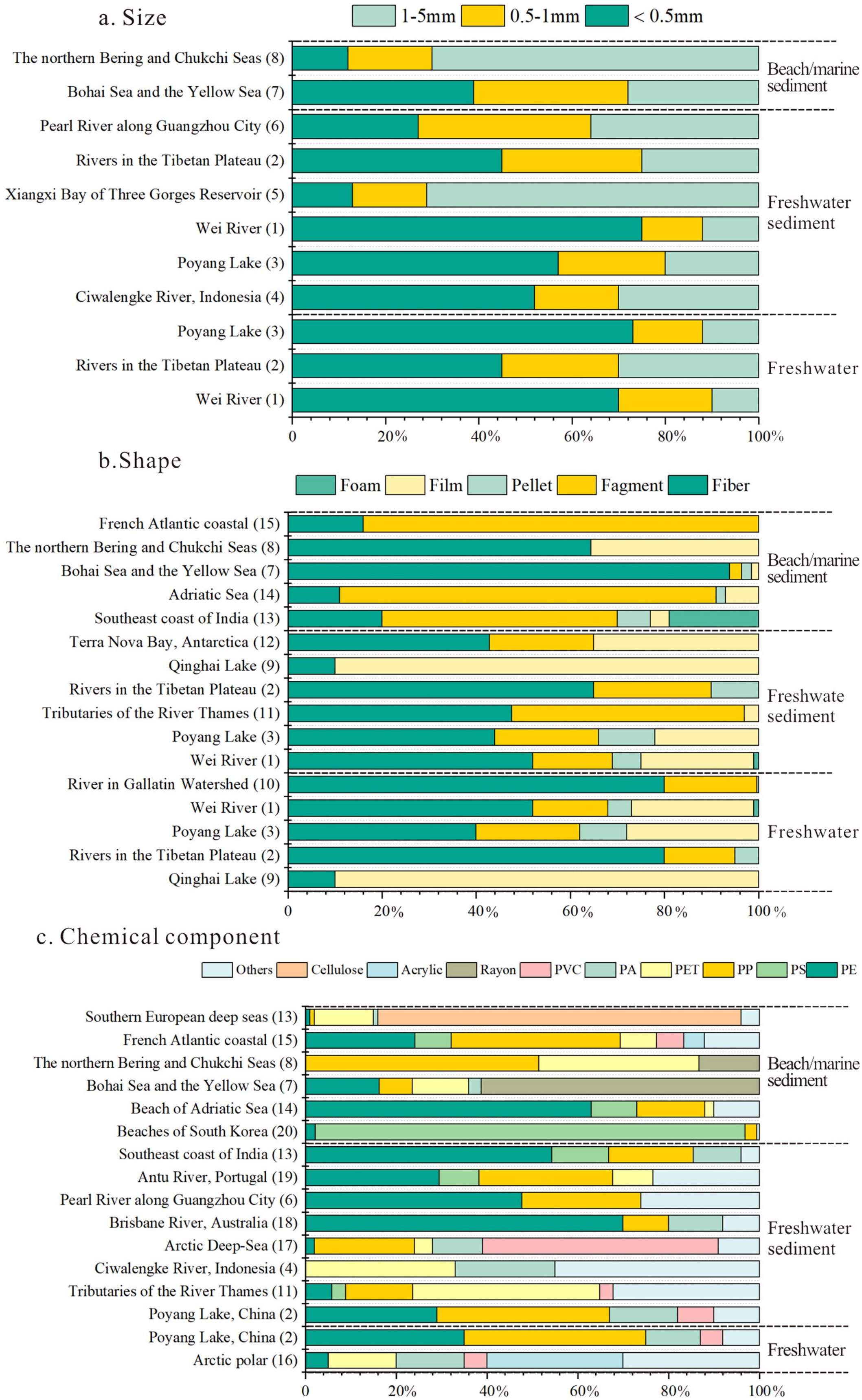

Fig. 3. A comparison with microplastics' size (a), shape (b), chemical component (c) in freshwater sediment with those from marine sediment and surface water of freshwater systems worldwide. References: 1. Ding et al., 2019; 2. Jiang et al., 2019; 3. Yuan et al., 2019; 4. Alam et al., 2019; 5. Zhang et al., 2018; 6. Lin et al., 2018; 7. Zhao et al., 2018; 8. Mu et al., 2019; 9. Xiong et al., 2018; 10. Barrows et al., 2018; 11. Horton et al., 2017; 12. Munari et al., 2017; 13. Karthik et al., 2018; 14. Vianello et al., 2018; 15. Phuong et al., 2018; 16. Lusher et al., 2015; 17. Bergmann et al., 2019; 18. He et al., 2020; 19. Rodrigues et al., 2018; 20. Eo et al., 2018.

This is also true for sediment samples from the ocean, the lower limit is usually between 50 and 333 μm and the mean size of microplastics seems to be larger than the pore size of sieve/filter.

Previous studies reveal that fiber and fragment account for the overwhelming majority in beaches and freshwater according to the morphological characteristics of microplastics (Andrady, 2017; Li et al., 2020; Yu et al., 2020). In the microplastics of freshwater sediments, fiber and fragment are also the most common shapes (Table 2). A considerable fibrous microplastics in freshwater come from clothing, blankets and other fiber products, and washing is an important pathway that releases them into the environment (Jiang et al., 2018; Lin et al., 2018). Fragmented microplastics could possibly originate from the exposure of larger plastic garbage to strain, fatigue, or UV light. The difference between the freshwater system and the marine system in the secondary source microplastics produced by environmental weathering is unclear (Eerkes-Medrano et al., 2015). Even for marine systems, the rate at which microplastics break and degrade is unknown (Cózar et al., 2014). Physical forces may be different to some extent. For example, storm and wave action in marine systems, while plastics in freshwater systems are more subject to physical and chemical degradation (Eerkes-Medrano et al., 2015; Free et al., 2014). Xiong et al. (2018) surveyed microplastics in the Tibet Plateau and suggested that particles may experience relatively high levels of weathering due to strong ultraviolet (UV) irradiation, windy, and dry (Xiong et al., 2018).

Different polymers have different densities, thus directly affect the way microplastics enter freshwater sediments (Zhang L. et al., 2020). The most common polymer types of microplastics found in the surface water of freshwater environment and beach samples are less dense polymers (PE, PP, and PS), and the densities of which are around 1.0 g/cm^3 (Andrady, 2017; Li et al., 2020). Similarly, PE, PP, and PS also account for the highest proportion of freshwater sediments. The abundance of these three polymer types are closely related to huge demand. The distribution of plastics demand by resin type in 2018 shows that PP, PE and, PS are the most in demand plastics in the world, among which the demand for PP and PE ranks the first and second in the world (Plastics Europe, 2019). PP, PE, and PS are mostly used in food packaging, reusable bags, and other plastic products with short life and disposability. Furthermore, the consistency of polymer types between beaches or marine sediments and freshwater systems may be an indicator of these interlinkages and source-pathways (Barrows et al., 2018; Zhang Y. et al., 2020). More recently, researchers believe that 80% of microplastics in the ocean comes from land (Jambeck et al., 2015) and rivers are one of the main ways for microplastics to reach the ocean (Eerkes-Medrano et al., 2015; Rochman, 2018).

5. Sources and factors affecting abundance of microplastics

5.1. Sources of microplastics

Microplastics are theoretically divided into two parts: primary microplastics and secondary microplastics. Primary microplastics refer to industrial products of plastic particles, which are discharged into the water environment through rivers and sewage treatment plants, etc. Microplastic particles contained in cosmetics or plastic particles and resin particles as industrial raw materials are typical primary microplastics. Microbeads were detected with an average of 20,860 particles/g in facial scrubs sold in China (Cheung and Fok, 2017), and it was estimated that an average of 209.7 trillion microbeads (306.8 t) are discharged into the water environment every year in China (Zhang et al., 2018). In the Riverbank sediments of the Rhine-Main River, the detected microplastic pellets with a particle size between 63 and 200 μm are closely related to the artificial microspheres used in cosmetics and detergents, which highlights the source of primary microplastics (Klein et al., 2015). As manufactured plastic microparticles produced in this size range are mainly applied to cleansers and cosmetics (Mendoza and Balcer, 2019; Zhang et al., 2018). The microbeads found on the St. Lawrence River sediment with small size and polyethylene composition confirm the source of consumer products (Castañeda et al., 2014). Leaking from small product factories is also a potential way for primary microplastics to be released to the environment (Mai et al., 2018; Peng et al., 2018). Artificial resin pellets with a regular shape and a diameter of 3–5 mm are used as raw materials in the manufacture of plastics, which also constitute the source of the primary microplastics (Hernandez et al., 2017; Lusher et al., 2017). However, microbeads and resin pellets are not often detected in the microplastics from freshwater sediments. One of the reasons could be the retention of primary microbeads in sewage sludge (Zhang et al., 2018).

Larger plastic products on land break into smaller particles when exposed to the elements until they eventually become microplastics (Auta et al., 2017; Cózar et al., 2014; Di and Wang, 2018). These microplastics are secondary microplastics, which are the main source of microplastics in the environment. The broken plastic products that are widely used in packaging, construction, electrical and electronic products, agriculture, automotive, and household products are an important source of microplastics (Li J. et al., 2018; Zhang et al., 2018). At the end of the life of plastic products, many of them are not recycled or incinerated but discarded into the environment by dumping (Li J. et al., 2018; Stolte et al., 2015). Under the combination action of several environmental factors (such as sunlight and temperature, biological effect), the weathering process slowly decomposes these products and generates a large number of secondary microplastics (Andrady, 2017; Li et al., 2020).

Many microplastic fragments are found in the Tibetan plateau, Nakdong River, and Lagoon of Venice, and the authors believe that these fragments are caused by the degradation and decomposition of larger household plastic products (Eo et al., 2019; Vianello et al., 2013). Cracks, pits, and attached particles on the surface of microplastic fragments are good evidence of these physical forces (Xiong et al., 2018). Among plastic products, the plastic package having the shortest lifespan is usually cheap and disposable. After use, many plastic packages are discarded into the environment and may produce a large number of film-like microplastics (Alimba and Faggio, 2019). Also, rubber particles that are generated by tire wear during pavement wear are also considered to be microplastics (Yukioka et al., 2020).

Fibrous microplastics in Bizerte Lagoon are considered to be discharged by wastewater from the textile industry, fishing gear, fishing nets, and shellfish farming materials (Abidli et al., 2017). The type of microplastics detected in the form of polyester, cotton, and nylon in Ciwalengke River generally result from clothing, which affirms the microplastics originated from broken cloth shredded cloth (Alam et al., 2019). Hernandez et al. (2017) simulates a quantitative experiment of microplastic fibers released from synthetic (polyester) textiles during the home wash. This study finds that thousands of fibers are discharged during the washing process, and the use of detergents would affect the total fiber mass, which provides strong evidence of the fibrous microplastic source. It has been predicted that up to 6000,000 fibers per 5 kg wash can be discharged in wastewater (Rodrigues et al., 2018). Fibrous microplastics can also be generated during routine use or wearing of textiles, so far, this part of the source has not been well documented (Zhang et al., 2018).

The types of microplastics have a certain spatial relationship with human activities (Cheung and Fok, 2017; Eerkes-Medrano et al., 2015). Primary microplastics are usually found in samples from many developed areas or the industrial area (Zhang et al., 2018). For instance, virgin pellets were found in the urban river of Shanghai, and there is a small plastic manufacturing plant nearby (Peng et al., 2018). Pellet was the most abundant in the sediment of the Ciwalengke River (Castañeda et al., 2014). The lack of primary microplastics but an abundance of fragments and fibers in the sediment of a very low resident population within rivers and lakes in the Tibetan plateau indicated originated from the crush or damage of household plastic products and garbage (Jiang et al., 2019; Xiong et al., 2018; Zhang et al., 2016).

5.2. Pathways for the transport of microplastics into inland sediment

The transmission of microplastics in freshwater environments is shown in Fig. 4. Household sewage is considered to be an important source of microplastics in freshwater, especially virgin plastic pellets/ microbeads and fibrous microplastics (Wong et al., 2020; Zhang et al., 2018). The wastewater treatment plant directly receives microplastics from landfills, industry, domestic wastewater, and rainwater (Mahon et al., 2017). Due to its small size, the settling operation of the sewage treatment plant can hardly remove the microplastics; rather, many of the microplastics are accumulated in the sludge. The presence of high concentrations of microplastics in the sewage disposal plant has been confirmed in many studies (Li X. et al., 2018; Mahon et al., 2017; Ziajahromi et al., 2017). Mahon et al. (2017) finds that the concentration range of microplastics separated from sludge is $4.2–15.4 \times 10^3$ particles/kg dry sludge (Mahon et al., 2017). Murphy et al. (2016) estimates that 2000 microplastics are released in the effluent from a sewage treatment plant with a population equivalent of 650,000 on one particular day, equivalent to 0.16 microplastic/person/day or 0.009 microplastic/ L (Murphy et al., 2016). In addition, due to the lack of or access to sewage disposal facilities and garbage disposal in remote areas, a large number of microplastics or potential microplastics generated from laundry, personal cleaning products are piled up or discharged into the river and eventually accumulate in freshwater sediments.

Rainwater runoff is another important way to transfer microplastics from the terrestrial environment to the inland water system (Allen et al., 2019; Zhang et al., 2018). Unregulated plastic garbage including domestic waste, agricultural plastics (such as plastic mulch and plastic woven bags) are usually scattered on roads, riversides, and fields or other unregulated dumping sites. The occurrence of rainstorms helps the plastic waste to be transferred to the freshwater system by rainwater runoff. Although this phenomenon is commonplace, more simulation and empirical evidence are needed for the data on the discharge and transportation of microplastics from rainwater runoff. Nizzetto et al. (2016) finds that microplastics with a particle size of less than 0.2 mm and a density less than water can be transferred from the catchment area to inland water systems and marine environments (Nizzetto et al., 2016). Apart from plastic wastes from littering, microplastics from tire and road wear also contribute to the source of microplastics for the transport of runoff. The microplastics originated from tire and road wear can reach 42% of all emissions exported by rivers to the seas (Yukioka et al., 2020).

5.3. Factors affecting abundance of microplastics in freshwater sediment

Many factors have been proposed to influence the abundance of microplastics in freshwater sediment. Factors that affect the abundance of microplastics include population density near water catchment, proximity to urban centers, water flow velocity, water catchment size, type of waste management used, and sewage spillage (Eerkes-Medrano et al., 2015; Klein et al., 2015; Zhang et al., 2018). Generally, the sampling sites are in areas with high human activity (high urbanization and high industrialization), it is expected that high pollution levels will be observed in sediments (Eerkes-Medrano et al., 2015; Rochman, 2018). The model is based on the proximity of microplastic sources. This theory has been proven in many studies. For example, microplastic pollution in the rivers of the Tibet Plateau finds that the abundance of microplastics in the sediment near Lhasa is relatively high and that in the sparsely populated areas is generally low (Jiang et al., 2019). Lhasa is the political, economic, cultural, scientific, and educational center of the Tibet Autonomous Region of China. The higher population and a large number of tourists are a reasonable explanation for the higher microplastics concentration here (Jiang et al., 2019). The spatial distribution of microplastics in the Three Gorges Reservoir shows that sampling sites with microplastics abundances above 5000 items/m^3 are almost located in densely populated urban areas (Di and Wang, 2018). Furthermore, high microplastics densities are recorded in the Taihu Lake, Pearl River, Wen-Rui Tang River, Urban recipient in Norway, and St. Lawrence River, where the rivers and lakes experience intensive industrial activity and tourism (Castañeda et al., 2014; Haave et al., 2019; Lin et al., 2018; Su et al., 2016; Wang et al., 2018).

However, Klein et al. (2015) found that neither population density nor industrial activity or the location of wastewater treatment plants seemed to be a good indicator of microplastic pollution in riverbank

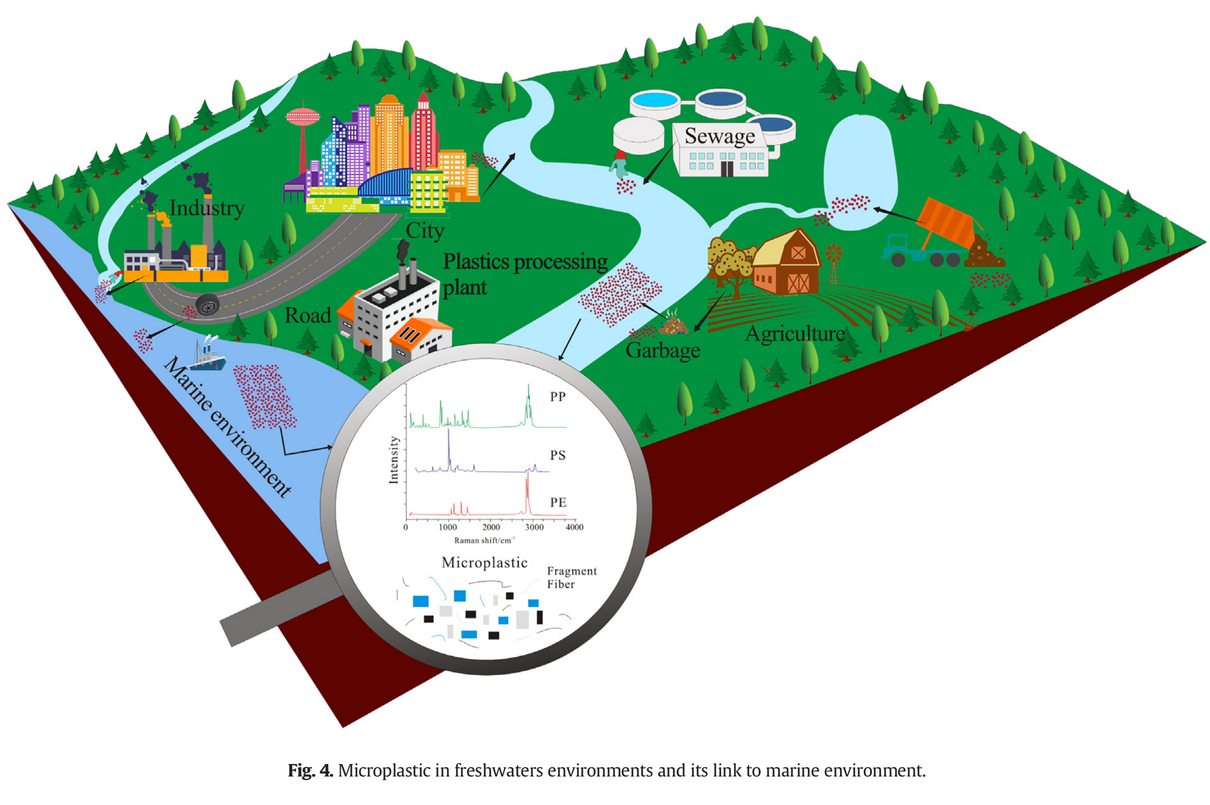

Fig. 4. Microplastic in freshwaters environments and its link to marine environment.

sediments in the monitored region (Klein et al., 2015). The possible reason for the lack of correlation between microplastics abundance and these factors may be that these factors are superimposed by the hydrodynamic forces (Alam et al., 2019; Li et al., 2020; Su et al., 2016). It is also important to note that population density does not represent the location of other point sources, such as industrial activities or wastewater treatment plants (Klein et al., 2015; Yuan et al., 2019). Moreover, no direct relation is observed between microplastic abundance in surface water and sediment samples in Poyang Lake and Three Gorges Reservoir (Di and Wang, 2018; Yuan et al., 2019).

5.4. Factors affecting the distribution of microplastics in the environment

The distribution of microplastics in freshwater sediments is still not completely clear, but the key to mastering its distribution is to understand the external forces driving its movement (Cózar et al., 2014; Law and Thompson, 2014). Quantitative and modeling methods are needed to simulate the effects of various physical forces on microplastic transport and diffusion on a range of spatial scales.

The distribution of microplastics in the marine environment demonstrated by quantitative and modeling methods indicates that large-scale forces such as wind-driven surface currents and geostrophic circulation have driven the diffusion model of microplastics in marine areas (Cózar et al., 2014; Eerkes-Medrano et al., 2015; Lusher et al., 2015). Moreover, at smaller spatial scales, experiments and field investigations have shown that the driving force of wind affects the spatial distribution of microplastics (Eerkes-Medrano et al., 2015). The global distribution of microplastics pollution in the open ocean summarized by Cózar et al. (2014) showed that microplastics accumulated in the convergence zones of five subtropical circulation areas, and the abundance was similar in each convergence zone (Cózar et al., 2014).

The spatial distribution of microplastics in the aquatic environment is formed by the interaction of the external force that drives force of large-scale and the characteristics (such as density, shape, and size) of microplastics, as well as the environment in which they are situated (Allen et al., 2019; Zhang Y. et al., 2020). Density is an important factor that affects the transportation of microplastics. The density of commonly used consumer plastic products is usually between 0.8 and 1.0 g/cm^3, while polymer types such as polyvinyl chloride (PVC), and polyethylene terephthalate (PET) have densities higher than that of water. Thus, the density of microplastic particles can roughly determine their storage space, distant seas, or benthic organisms; low-density microplastic particles tend to occupy the surface of the aquatic environment, while high-density microplastics are more likely to appear in deep seas and benthic organisms (Eerkes-Medrano et al., 2015). Biological fouling and adsorption of other contaminants can affect the size and density of microplastics, thus affecting their distribution and diffusion.

Depending on the characteristics of suspended sediments, the driving factors affecting the migration and diffusion of microplastics from freshwater sediments are not only related to the characteristics of microplastics themselves but also controlled by environmental factors such as water depth, flow velocity, matrix type, bottom topography and seasonal variability in water flow (Conkle et al., 2018; Eerkes-Medrano et al., 2015). In Wei River, the highest abundance was found in a large wetland park with a very low flow velocity. Declining river velocity makes it easier for microplastics to settle, which is responsible for the high concentration in the area (Ding et al., 2019). Storm events and flood events also seem to have a temporal aspect in the abundance and distribution of microplastics in sediment. In Antuã River, the abundance of microplastics in sediments has obvious seasonality. The highest abundance of microplastics presents at the end of the rainy season, while the lowest abundance of microplastics occurs at the end of the dry season (Rodrigues et al., 2018). Other physical factors may also have a temporal impact such as the tidal cycle in estuaries and dam release (Stolte et al., 2015; Zeng, 2018; Ziajahromi et al., 2017).

6. Effects and potential risks of microplastics in freshwater sediment

In the aquatic environment, microplastics can have direct and indirect effects based on their adsorption of toxic substances, as well as the access to be absorbed by various organisms (Zhang Y. et al., 2020), which make microplastics a hazardous contaminant and present risks to ecosystems. Furthermore, the persistence and ubiquity of microplastics highlights the complexity of potential risks.

6.1. Microplastics and their chemical components

The components of microplastics such as monomers and additives, can be released during the use and disposition of products, some of which may be harmful to the environment (Yu et al., 2020). Plastic is composed of synthetic resin and filler, plasticizer, stabilizer, lubricant, colorant, and other additives. The monomer is the basic unit of plastic polymer, and studies have revealed that some monomers are harmful to humans. According to the polymer hazard rating model established by Lithner et al. (2011), polystyrene is listed as one of the most dangerous polymers mainly because the monomer that produce polystyrene has the risk of monomer mutagenesis or carcinogenesis (Lithner et al., 2011; Yu et al., 2020). Styrene has been listed as a carcinogen by the International Agency for Research on Cancer. In any case, styrene can be evolved during polystyrene weathering as recently reported (Lomonaco et al., 2020). Except for the monomers, several of the common plastic additives such as flame retardants, plasticizers, heat stabilizers, and antioxidants are considered to be harmful. For example, plasticizers can disrupt animal endocrine systems. Heat stabilizers seem to produce toxic effects on the environment, especially biological organisms. In a word, microplastics and their chemical components present certain health risks (Fahrenfeld et al., 2019; Xu S. et al., 2020).

The increased surface area of the microplastic compared to plastic matrix enhances the adsorption of chemical contaminants present in the surrounding environment (Hale et al., 2020), including persistent organic pollutants (Heskett et al., 2012), antibiotics (Yu et al., 2020), and heavy metals (Jasna et al., 2018). Different types and shapes of polymers have different adsorption of pollutants (Rochman et al., 2019). Qu et al. (2018) simulated found that venlafaxine and its metabolite *O*-desmethylvenlafaxine can be adsorbed by PVC microplastics up to 80% (Qu et al., 2018). High levels of persistent organic pollutants were sporadically found in plastic pellets collected from remote islands (Heskett et al., 2012; Jasna et al., 2018). Jasna et al. (2018) investigated the degree of pollution of microplastic pellets on the beaches of Vis island and found that the trace metals concentrations in microplastics pellets were greater than that of reported in seawater, which revealed that microplastic pellets sorb trace metals from the marine environment (Jasna et al., 2018).

As a vehicle, microplastics may carry the adsorbed trace metals enter the food chain due to incidental ingestion of microplastic particles by marine animals, which might magnify the bioaccumulation of contaminants (Jasna et al., 2018; Qu et al., 2018). While destroying the aesthetic value of the water environment (Fahrenfeld et al., 2019; Li X. et al., 2018), microplastics are likely to pose threats to public health and cause biodiversity loss (Cózar et al., 2014; Elert et al., 2017). Therefore, exposure of microplastics in the aquatic ecosystem needs further in-depth investigations to evaluate their environmental risks more accurately.

6.2. Effects of microplastics on organisms

With the decrease in size of microplastics, they can be ingested by a wider variety of organisms (Cózar et al., 2014; Prokic et al., 2019). Evidence of the impact of microplastics intake on freshwater systems is relatively few, and most of them recorded in the ocean. (Eerkes-Medrano et al., 2015). The recent findings of microplastics in organisms highlight the effects of mechanical damage to the gut of feeding organisms, digestive tract obstruction, slow metabolism, and reduced fertility (Tang et al., 2018; Wright et al., 2013; Zhang et al., 2018).

Various studies implied that the effects of microplastics on organisms are related to their chemical composition, size, and shape. The opportunity for the aquatic organisms to encounter or uptake microplastics is linked to the two characteristics of microplastics: size and density (Van Cauwenberghe et al., 2015b; Zhang et al., 2018). For example, ingestion and entanglement of microplastics for benthic suspension and deposit feeders may be more likely to occur in particles with a density higher than that of freshwater and seawater (as they sink to the underwater and seafloor) (Rochasantos and Duarte, 2015). As long as the particle size of the microplastics is close to that of the sediments or even smaller, the microplastics can be ingested not only by low-trophic organisms but also by other benthic organisms (Wong et al., 2020).

It has been estimated that approximately 690 species were affected by marine plastic pollution in 2015, with at least 10% of the species ingesting microplastics (Yu et al., 2020). Furthermore, microplastics have been recorded in the guts or tissues of many aquatic organisms, including zooplankton (Sun et al., 2017, 2018), bivalves (Van Cauwenberghe et al., 2015b), and fish (Akhbarizadeh et al., 2018; Lusher et al., 2017). Plastic or microplastic debris have a direct mechanical effect on aquatic organisms through entanglement and swallowing (Su et al., 2018; Yu et al., 2020). These mechanical wear and tear on the digestive tract reduce the food intake of aquatic organisms, and eventually lead to starvation and death. Toxicological hazards mainly come from additives in microplastics and chemical substances adsorbed on the surface of microplastics (Akhbarizadeh et al., 2018; Yu et al., 2020). These toxic chemicals and additives enter the organism along with microplastics and can be released during the desorption process, thereby causing toxicological effects on the organism (Kirstein et al., 2016).

Microplastic intake has been recorded from zooplankton to large mammals (Cole et al., 2015; Xu S. et al., 2020). Zooplankton connects primary producers with higher nutritional levels and therefore plays an important role in the marine food chain (Sun et al., 2017). High concentrations of microplastics were detected in zooplankton in the northern South China Sea and East China Sea (Sun et al., 2017, 2018). The ingestion of microplastics by zooplankton provides important access to the marine food chain, which transfers microplastics to higher nutritional levels along the food chain. The transfer of microplastics between mussels and crabs through nutrient transfer has been confirmed and microplastics can be distributed in the hemolymph and tissues of crabs (Farrell and Nelson, 2013).

There are relatively few studies on freshwater organisms. Andrade et al. (2019) provided the first evidence of plastic polymers ingestion by freshwater fishes in the Amazon (Andrade et al., 2019). In a major tributary of the lower Amazon, the examination of the stomach contents of the three trophic guilds of fish (herbivore, omnivore, and carnivore) found that approximately 80% of the species analyzed had plastic particles, and the length of plastic particles was in the range of 1 to 15 mm (Andrade et al., 2019). Furthermore, microplastics were even found in the digestive tracts of fish in remote Tibetan Plateau regions with the concentration varied from 2 to 15 items per individual (Xiong et al., 2018). Fossi et al. (2017) simulated and found that a high occurrence of microplastics spatial distribution consistent with the distribution of marine plastic litter, which highlights the risk of fin whale exposure to microplastics (Fossi et al., 2017).

In the investigation of microplastics and heavy metals concentration in muscles of four commercial fish in southwest of Iran, microplastics with variety of shapes, colors, and size were found in all investigated fish muscle samples and the mean concentration of microplastics in the muscles of studied fish were 8.00 ± 1.22 and 7.75 ± 2.16 items/10 g fish muscle, respectively (Akhbarizadeh et al., 2018). The chemical toxicity of microplastics and heavy metals shows a good linear relationship in some species, which reveals a health threat to consumers. The microplastic pollution of benthic fish is even more worrying, especially those that live on and feed on sandy and muddy seabeds. Thus, the continuous find of microplastics in human feeding organisms makes the study on the direct and potential harm of microplastics to humans very urgent.

7. Conclusions

Microplastics have become one of the emerging pollutants in the aquatic environment. The occurrence of microplastics in rivers, lakes, and reservoirs has continued to expand on a global scale, which has attracted widespread attention from scientists, policymakers, and the public. However, there is still insufficient knowledge about the rapid monitoring, distribution, and influencing factors of microplastics in freshwater sediment.

Among published studies, investigations on microplastics in the sediment of river and lake are intensive and detailed, while relatively little attention is paid to that of reservoirs. The relative abundance of microplastics in freshwater sediments exhibited various characteristics and quantities in different regions. The fiber was the most common shape for microplastic in freshwater sediments. Smaller microplastics were found to be more abundant in freshwater sediment, with particle sizes less than 1 mm. The most common colors observed in freshwater sediments were white and transparent. For microplastics in freshwater sediment, PE is the main polymer, followed by PP and PS, and the chemical composition varies greatly in different regions. Secondary microplastics derived from larger plastic products on land constitute the main source of microplastics in freshwater sediments. Due to its multi-source nature, neither population density nor industrial activity or the location of wastewater treatment plants seemed to be a good indicator of the spatial distribution of microplastics pollution in freshwater sediment. Hydrodynamic conditions, rainfall, and flood events superimposed effects on the spatial distribution of microplastics in freshwater sediment.

Since the study of microplastics in freshwater sediments is still in its infancy, there are still some inconsistencies in the description and comparison of microplastics abundance and characteristics. Standardization methods for sampling and measurement of microplastics in freshwater sediment are imminent. Worldwide study on spatial and temporal variations of microplastics in freshwater sediment needs to be further enhanced, but the key is to understand the external forces driving its transport and diffusion.

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.

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