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## Microplastic pollution of worldwide lakes

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10 Abstract: Studies on microplastic (MP) pollution in lakes are recent, although the problem of  
11 MP particles in the oceans was first discovered in the 1970s. The first study on lakes was  
12 published in 2011. Since then, to our knowledge, 98 lakes have been investigated worldwide.  
13 In recent years, studies on this topic have increased worldwide, particularly those focusing on  
14 urbanised lakes. Most of the plastic waste in the seas and oceans originates from the terrestrial  
15 environment and inland waters. Moreover, lakes are potential temporary or long-term MP  
16 accumulators, according to the residence time of water. They are also of high interest for  
17 biodiversity, ecology, and the economy. Lacustrine ecosystems may suffer the same fate as  
18 marine ecosystems, or even worse, owing to their greater exposure. With the significant focus  
19 on ocean and sea contamination, contamination of freshwater ecosystems and lakes is a new  
20 and rising topic. However, as a new field of research, several methodological issues have been  
21 raised. The team diversity worldwide has led to contrasting sampling techniques and  
22 materials, sample treatments, analyses, and presentation of results. Consequently, it is

23 necessary to determine several consensuses between scientific teams in order to work together  
24 with accuracy, produce comparable results, speed up knowledge sharing and reduce the  
25 reproducibility crisis. This review focuses on (1) MP contamination in 98 worldwide lakes.  
26 We identify (2) the theoretical sources of MPs and provide (3) an estimate of MP pollution in  
27 different compartments of the lakes based on current state-of-the-art methods. In addition, we  
28 also report (4) the predominant MP size classes and polymer types. Finally, we suggest (5)  
29 several recommendations to build a consensus between all the working teams to facilitate  
30 decision-making by public authorities.

31 Microplastic pollution has been investigated in 98 lakes around the world by using different  
32 methods and materials which leads to incomparable data.

## 33 1. Introduction

34 Macroplastics, microplastics (MPs), and nanoplastics have been widely studied in the past  
35 few years, particularly in marine and oceanic environments. Plastic pollution in aquatic  
36 environments was observed for the first time in the 1970s. In the Sargasso Sea, the plastic  
37 concentration was close to 3,500 items/km<sup>2</sup>, which was mainly comprised of cigarette holders  
38 (Carpenter and Smith, 1972). Recently, many studies have focused on plastic pollution in  
39 freshwater environments, especially rivers and lakes. In 2011, 80 % of the plastic pollution in  
40 seas and oceans originated from the terrestrial environment (Andrady, 2011). In 2019, the MP  
41 contamination observed in the terrestrial environment was 4 to 23 times higher than that in  
42 marine ecosystems (Horton et al., 2017). Freshwater is considered one of the sources of MPs  
43 flowing into the seas, and rivers are considered the main transport vector of plastic debris  
44 from terrestrial sources (Iannilli et al., 2020). It is of high importance to study freshwater  
45 ecosystems and analyse the plastic pollution sources, dispersion dynamics, accumulation, and  
46 fate of these pollutants. Lacustrine ecosystems are pollutant accumulators, and consequently,  
47 indicators of catchment functioning. Climatic, geographical, and physical characteristics  
48 determine the residence time of water in lakes. Pollutant circulations depend on this duration;  
49 a short retention time leads to quick contamination and cleaning, whereas a long retention  
50 time leads to slower changes in the plastic contents and additional processes such as local  
51 fragmentation (Montuelle et al., 2015). Therefore, lakes with long water retention times seem  
52 to accumulate a greater amount of MPs, as observed in Mongolia (Free et al., 2014; Iannilli et  
53 al., 2020). Some plastic polymers are positively buoyant and float on the surface or strand on  
54 the coast; some are negatively buoyant and will remain in the water column or sink to the  
55 benthic zone. The behaviour of polymers plays a crucial role and influences the subsequent  
56 contamination of rivers, seas, and oceans. Furthermore, human societies are dependent on  
57 these ecosystems. Lakes offer many resources to societies based on ecosystem services such

58 as control services (flooding and drought), provision services (drinking water; food  
59 production via fisheries, aquaculture, and irrigation of agricultural lands; and energy  
60 production via hydropower dams), cultural services (recreation), and support for favourable  
61 conditions for sustainable nutrient cycling. Lakes are essential to the function of the economy  
62 and offer many ecosystem benefits. Halieutic resources allow the local development of  
63 professional fishing spots, restaurants, and culinary businesses; sports and recreational  
64 possibilities are key elements for tourism; and biodiversity contributes to good ecological  
65 functioning and reinforces tourism. In addition, MPs, which threaten fauna (Bellasi et al.,  
66 2020), could endanger humans through their diet (Cox et al., 2019; Koelmans et al., 2019; Q.  
67 Zhang et al., 2020), and may be vectors of potentially pathogenic bacteria and viruses (Arias-  
68 Andres et al., 2018; Kirstein et al., 2016; Wu et al., 2019). Currently, MPs are ubiquitous in  
69 urbanised and more isolated environments. In 2010, worldwide plastic production reached  
70 265 million t (PlasticsEurope, 2011). In 2018, it reached 359 million t (PlasticsEurope, 2019).  
71 In the same year, European plastic production reached 62 million t (PlasticsEurope, 2019).  
72 According to a recent study, the annual production of plastic fibres increased by more than 6  
73 %/y, reaching 60 million t in 2016, or approximately 16 % of global plastic production  
74 (Gasperi et al., 2018). In 2019, it was predicted that the proportion part of poorly managed  
75 plastic waste could increase, from 60-99 million t in 2015 to 155-265 million t by 2060  
76 (Lebreton and Andrady, 2019). If this estimation is correct, then lacustrine ecosystems will be  
77 significantly affected by MP contamination. However, knowledge of plastic pollution in lakes  
78 worldwide is still incomplete. In 2012, the mean publication rates on MP pollution were: 23,  
79 5, and 2 papers for marine, freshwater, and estuarine ecosystems, respectively (Meng et al.,  
80 2020). Nevertheless, studies concerning MP pollution in lakes have recently increased. This  
81 review examines all the studies concerning MP in lakes to date to collect available data and to  
82 understand the types of plastic pollution in the lacustrine ecosystem. MP pollution has been

83 studied in 98 lakes worldwide (**Tables S1- S2**). The aim of this work was to compare the  
84 worldwide MP contaminations in lakes. Unfortunately, in view of the high heterogeneity of  
85 the sampling protocols, result reports, and definitions used, comparisons among the obtained  
86 results are laborious. Therefore, we carefully compared: (1) the methods to investigate MP in  
87 lakes and we suggested a normalised protocol, (2) all the MP contaminations in the various  
88 environmental compartments of each lake in regards with methodologies and sampling  
89 protocols and, (3) MP contaminations sources according to demographic pressures (urban vs  
90 rural lakes).

## 91 2. Material and methods

### 92 2.1. Sampling methodologies

93 Among all the studies investigated MPs in lakes, three sampling methods have been used: net  
94 was the most frequent (22 articles), pump (7 articles) and, sampler/jar/bucket are more  
95 occasionally employed (8 articles taken together). Few studies were using pump system  
96 because of its cost and specific purpose (net is also used in plankton studies); however, it  
97 allowed to easily filter different MP size classes compared to net system. Two major  
98 difficulties were related to pump sampling: i) to convert a precise measurement of the filtered  
99 volume ( $m^3$ ) into a filtered surface to provide a density ( $MP/km^2$ ) for floating MP and ii)  
100 avoid clogging of a limited filtration area while filtering large volume of water to insure the  
101 sampling representativeness. The use of buckets, jars and samplers also led to volume  
102 limitation due to the necessity to take the samples back to the laboratory and consequently,  
103 one could question the sample representativeness. However, filtration conditions in the  
104 laboratory would be better controlled. Net systems were most common (22 articles) because  
105 they can also be used in plankton studies and usual mesh size ( $300\ \mu m$ ) were fairly low cost  
106 (nets  $< 300\ \mu m$  are more expensive). Unlike pump methods, nets have a large filtration area

107 which limits clogging and allows larger and thus more representative samples. Nevertheless, a  
108 major bias in MP investigation using nets lied in the difficulty to properly assess the filtered  
109 volume. Indeed, authors assumed that water velocity within the net equal the boat speed or the  
110 flow velocity in the river which was incorrect and induced an overestimation of the filtered  
111 volume and a subsequent underestimation of the MP contamination. For instance, (Gateuille  
112 et al., 2020) reported an averaged filtration velocity, three times lower than the boat speed  
113 using a 50  $\mu\text{m}$  net. Therefore, authors willing to use nets should be able to accurately measure  
114 the flow velocity in the net. The measurement of the instantaneous filtration velocity is also  
115 an efficient tool to monitor the net clogging and to stop the sampling as soon as the net  
116 clogging starts therefore preventing a surface water backflow that could lead to the loss of  
117 floating MPs. Differences in results were observed when using a net or pump (Uurasjärvi et  
118 al., 2020). The MP concentrations in the surface water of Kallavesi Lake (Finland) greatly  
119 differed according to the sampling strategy (even if stations were not similar between the net  
120 and pump samples), with  $0.27 \pm 0.18$  particles/ $\text{m}^3$  obtained with a 333  $\mu\text{m}$  Manta net and  $1.80$   
121  $\pm 2.30$  particles/ $\text{m}^3$  obtained with a 300  $\mu\text{m}$  filtering pump (Uurasjärvi et al., 2020). However,  
122 the pump and net samplings were not carried out exactly at the same places during this study  
123 and the differences in MP concentrations could be due to spatial heterogeneity rather than  
124 sampling protocols. Therefore, further investigations are required to accurately compare these  
125 two sampling methods (pump and net). As most of the published results were obtained using  
126 net sampling, and because of the ease of access of this device, we suggest the use of nets for  
127 surface water sampling, with the measurement of water velocity in the net to obtain a precise  
128 value of filtered water volume. Moreover, it is essential to reach an agreement on the mesh  
129 size (**Table S2**). The 300  $\mu\text{m}$  net is the most commonly used (particularly 300, 333, and 355  
130  $\mu\text{m}$ ). Approximately 80 % of 50 published aquatic surveys concerning MPs in aquatic  
131 environments only accounted for plastics  $\geq 300$   $\mu\text{m}$  in diameter, whereas > 95 % of particles

132 in surveyed products were smaller than the 300  $\mu\text{m}$  minimum diameter (Conkle et al., 2018).  
133 The use of 50  $\mu\text{m}$  or at least  $\leq 100 \mu\text{m}$  nets will reduce particle underestimation and fill the  
134 knowledge gap on the smallest plastic particles. Some researchers have already started  
135 working on smaller particles in lake surface waters, such as 20  $\mu\text{m}$  (Kaliszewicz et al., 2020),  
136 20  $\mu\text{m}$  (Bharath K et al., 2021), 38  $\mu\text{m}$  (Alfonso et al., 2020b), 50  $\mu\text{m}$  (Pastorino et al., 2020)  
137 and, 60  $\mu\text{m}$  (Bertoldi, 2020). In order to consider as many MPs as possible, the lowest mesh  
138 size should be used to avoid underestimation. However, using thinner nets can be difficult if  
139 there is a large number of suspended solids in the water. We suggest using nets with a mesh  
140 size between 50  $\mu\text{m}$  and 300  $\mu\text{m}$  to reach a compromise between the lowest amount of  
141 clogging and the highest quantity of sampled MPs. The sampling depths for surface waters  
142 were very heterogeneous, namely 16 cm, 20 cm, 30 cm, 50 cm and, 100 cm below the surface.  
143 The sampling depth was often not mentioned at all. The researchers working on MP in  
144 freshwaters (lakes, tributaries) should mention it to help readers to understand the results. In  
145 order to standardise studies, we suggest to sample water between surface and  $\sim 50$  cm at the  
146 most and to use another system to sample the column water independently. Further studies  
147 should focus on column water (from the bottom to surface water) sampling with a Niskin  
148 bottle or net lifting. This method would help to determine and understand the distribution of  
149 particles throughout the water column and their transfer to sediments according to lake  
150 characteristics. Finally, we advise scientific community to be rigorous in writing and  
151 explaining their sampling and laboratory techniques (e.g.: MP detection limit according to the  
152 analysis system, class size studied according to sampling methods). In order to ease the  
153 information research in a paper we suggest that authors provide with the main details of the  
154 methods (sampling, treatment, analysis) rather than only refer to scientific studies which is  
155 time consuming for the reader and sometimes refers to inaccessible publications. In addition,  
156 we suggest to sample the largest possible volume for each water sample to be more

157 representative and ease comparison of future scientific studies. Concerning the sediment  
158 compartment, it is important to describe the type (sandy, gravelly, or muddy) and the organic  
159 carbon content because of their influence on plastic bonds and mechanical pit capacities  
160 (Zbyszewski et al., 2014). In addition, the sample depth and thickness should be mentioned  
161 each time. We also recommend that authors report contaminations either in MP/m<sup>3</sup> or MP/kg  
162 (dry weight: dw or, wet weight: ww) with the corresponding sediment density. The studied  
163 samples should be precisely named following our suggestions (lake bottom sediments  
164 (LBoS), lakeshore sediments (LShS) or lake beach sediments (LBeS)) to help in cross  
165 analysis. LBoS and LShS are both under water, whereas LBeS is mainly a temporary sink for  
166 buoyant particles and can also be contaminated by direct discharge of MPs and macroplastics.  
167 Among the studies, different terminology was used, which led to misunderstandings.

## 168 2.2. Quantification methods, reporting of result, and definitions

169 First, we suggest using a shared format to report the MP concentrations in surface water,  
170 sediment, and fauna to ease the comparison between studies. Further investigations should  
171 study the MP concentrations by volume (particles/m<sup>3</sup>), the average particle size (ideally the  
172 volume or at least the minimum and maximum dimensions of the particle already provide a  
173 good indication), mass (particles/kg), and individuals (particles/aquatic organism) for water,  
174 sediment, and fauna, respectively. Moreover, it is essential to report at least the mean  
175 concentration and standard deviation. In addition, the minimal and maximal values could be  
176 provided. Further studies should always include the total and blank concentrations in parallel  
177 and not subtract the blank concentration to avoid the use of different subtraction methods.  
178 Second, it is important to precisely quantify the types and forms of polymers (percentage of  
179 the concentration) and avoid quantification ranking to better compare all the studies.  
180 Concerning the polymer types, several acronyms have been used among studies, thereby

181 making the parallel comparison more complex. For example, polyacrylate can be named PAK  
182 or PAC. We suggest naming polymers PE, PP, copolymer (CP), PS, nitrocellulose (NCE),  
183 PA, PET, PVC, EPS, PES, PAC (polyacrylate), polyurethane (PU), polycarbonate (PC),  
184 polymethyl methacrylate (PMMA), polybutyl methacrylate (PBMA), acrylonitrile butadiene  
185 styrene (ABS), styrene butadiene rubber (SBR), and others/unknowns (O/U). Copolymers can  
186 include PU-PET, PVC-PSS (polystyrene sulfonate), PE-PP, PE-PET, PAC-PET, VC-VAC  
187 (vinyl chloride/vinyl acetate), and ethylene-vinyl acetate (EVA). We suggest including natural  
188 and synthetic sources in cellulose given the complexity of discriminating between parental  
189 and regenerated compounds through spectroscopic analyses (Binelli et al., 2020; Hartmann et  
190 al., 2019; Stark, 2019). The main components of the tire particles (copolymers ABS and SBR)  
191 are not included given the complexity of identifying these particles by FT-IR spectroscopy.  
192 However, they could be a major source of MPs in the lacustrine environment. It is necessary  
193 to develop an efficient analytical methodology to precisely quantify these copolymers.  
194 Concerning the MP shapes, we suggest reporting them as follows to avoid ambiguous  
195 communication and generation of incomparable data: fibres (which includes fibres, filaments  
196 and lines), fragments, films/sheets, pellets (which includes granules, flakes, microbeads and,  
197 spheres) and, foams (which includes Styrofoam). Some studies differentiate fibres from  
198 filaments, pellets from granules but as it is difficult to visually differentiate them, combining  
199 them will avoid incorrect identification. Further studies should quantify the particle shapes by  
200 percentage (as polymer types) to expose differences among lakes worldwide and precise  
201 percentage estimates (by mass and/or number). Ideally, the current technological tools allow  
202 us to have the average size distribution of the particles which must be informed. If it is not  
203 possible, to easily compare the report of precedent and future results, we suggest using the  
204 same four size classes in further studies as follows: < 50  $\mu\text{m}$ , 50  $\mu\text{m}$ -300  $\mu\text{m}$ , 300  $\mu\text{m}$ -1 mm,  
205 1 mm-5 mm. Moreover, the detection limit of FTIR is approximately 6.25  $\mu\text{m}$ , we suggest to

206 set the limit at 8 times the measurement limit. We suggest a limit of 50  $\mu\text{m}$  because studies  
207 that worked on small plastic particles were at or below this size (Section 5.1), including 38  
208  $\mu\text{m}$  (Alfonso et al., 2020b), 45  $\mu\text{m}$  (Jiang et al., 2018), 47  $\mu\text{m}$  (Alfonso et al., 2020a), and 50  
209  $\mu\text{m}$  (Wang et al., 2017; Xia et al., 2020; Yuan et al., 2019), for surface water. Several studies  
210 worked at or above the 100  $\mu\text{m}$  limit for surface water, namely 100  $\mu\text{m}$  (Bordós et al., 2019),  
211 106  $\mu\text{m}$  (Cable et al., 2017), 112  $\mu\text{m}$  (Xiong et al., 2018), 125  $\mu\text{m}$  (Grbić et al., 2020), and  
212 150  $\mu\text{m}$  (Migwi et al., 2020). The 300  $\mu\text{m}$  frontier corresponds to the size of a plankton net,  
213 and so far, most studies have worked with 300  $\mu\text{m}$  or 333  $\mu\text{m}$  nets. Among studies concerning  
214 MPs in lakes, several quantified particles up to 999  $\mu\text{m}$  (Eriksen et al., 2013; Free et al., 2014;  
215 Mason et al., 2020, 2016) or 1 mm (Alfonso et al., 2020b, 2020a; Di and Wang, 2018; Oni et  
216 al., 2020). We suggest a consensus of 1 mm because most of the studies were interested in  
217 particles smaller than 1 mm and some define MP as particles below or equal to 1 mm. Finally,  
218 we suggest a limit of 5 mm according to the initial MP definition (Arthur and Bamford,  
219 2009).

### 220 2.3. Study sites

221 To our knowledge, MP contamination has been studied in 98 lakes to quantify the  
222 concentration or identify the shape and polymeric composition of plastic debris. However, to  
223 better understand the fate of MPs in lakes, researchers should consider and highlight  
224 numerous physical and environmental characteristics of freshwater ecosystems, especially the  
225 length of the main tributary (km) and its flow ( $\text{m}^3/\text{s}$ ), the water retention time (year), the  
226 surface area ( $\text{km}^2$ ), volume ( $\text{km}^3$ ), mean and maximum depth (m), watershed area ( $\text{km}^2$ ), lake  
227 altitude (m), land uses (percentage of urban, agricultural and road areas), the waste and  
228 wastewater treatment policy, population density (inhabitants/ $\text{km}^2$ ), and distance of inhabited  
229 areas to the lake (km), as shown in a recent study in **Table SI** (Alfonso et al., 2020b). These

230 environmental parameters will allow readers to understand the place of the lake, its  
231 environment, its watershed and the potential MPs sources that may be linked to it. Moreover,  
232 these provided characteristics will ease the comparisons between studies and the interpretation  
233 of the differences among results. Currently, there is almost no available data for remote lakes  
234 (**Tables S1- S2**), which is defined as an ecosystem far from any city, town, village, or hamlet  
235 that is only reachable through hiking and does not contain anthropogenic settlements (except  
236 for mountain huts) in the catchment basin. Remote lakes could also be affected by MPs via  
237 atmospheric fallout on the water surface or on the soils in the watershed. Investigation of  
238 these remote sites can help to quantify the flux of atmospheric deposition after mid-range or  
239 long-range atmospheric transport of MPs. It will also help to identify the impacts on aquatic  
240 life because of the high sensitivity of these ecosystems to xenobiotic pressure.

### 241 3. Estimation of microplastic pollution in worldwide lakes

242 The concentrations of MPs and other anthropogenic particles in Lake Ontario streams were  
243 correlated with urbanisation in the region (Grbić et al., 2020). In order to better compare the  
244 MP contamination between lakes subjected to various pollution pressures, we differentiated  
245 the lakes according to the proximity of urbanised areas and the number of inhabitants. We  
246 presumed that lakes distant to urbanised areas were more affected by atmospheric fallout and  
247 agricultural runoff than those close to densely inhabited areas, which were more affected by  
248 macroplastic pollution, impermeable surface runoff, and high-volume wastewater discharge.  
249 We distinguish “rural” and “urban” lakes as defined in Sections 2.1 and 2.2, respectively.  
250 MPs contamination on surface waters and sediments of rural and urban lakes were compared  
251 (**Fig. 1- 2**) according to size range studied, units (MP/m<sup>3</sup>, MP/km<sup>2</sup>, MP/kg and, MP/m<sup>2</sup>) and  
252 available data. Concerning surface waters (**Fig. 1**), values were comprised between 0.27  
253 MP/m<sup>3</sup> and 34,000 MP/m<sup>3</sup> with a median value around 1,442 MP/m<sup>3</sup> and, between 10,900

254 MP/km<sup>2</sup> and about 4,700,000 MP/km<sup>2</sup> with a median value of 48,200 MP/km<sup>2</sup>. We noticed the  
255 lack of available data on MP pollution of rural lakes. Indeed, only three values were available  
256 for rural lakes approximately 1 MP/m<sup>3</sup>, 4,000 MP/m<sup>3</sup> and, 20,000 MP/km<sup>2</sup> confirming the  
257 importance of investigating more rural and remote lakes. MPs in water showed larger size  
258 ranges for MP/m<sup>3</sup> than MP/km<sup>2</sup>, these differences in size ranges were linked to the sampling  
259 (e.g.: net, pump) and laboratory techniques (e.g.: filtration mesh). In some cases, large size  
260 ranges seemed to be in correlation with higher MP pollutions. However, differences in MP  
261 contaminations between lakes cannot be explained by methodologies variability only.  
262 Identifying the MP sources and the watershed characteristics could explain these differences  
263 as suggested in Section 1.3. Concerning sediments (**Fig. 2**), values were comprised between  
264 0.7 MP/kg to 4,220 MP/kg with a median value around 385 MP/kg and, between 17 MP/m<sup>2</sup>  
265 and 2,500 MP/m<sup>2</sup> with a median value around 290 MP/m<sup>2</sup>. Sediments in rural lakes could be  
266 less contaminated than urban ones. However, the variability of size ranges studied avoid an  
267 accurate comparison of the results. Most of MP contaminations in urban lakes were higher in  
268 surface waters and sediments than rural lakes. However, the lack of available data on MP  
269 pollution of rural lakes, the differences in methodologies and the lack of watershed data  
270 prevented us suggesting conclusions.

### 271 3.1. Rural lakes

272 We define a rural lake as a lake close to a hamlet(s), village(s), or town(s) (total  $\leq$  100,000  
273 inhabitants) ( $\leq$  10 km). In the current scientific literature, plastic contamination in different  
274 compartments has been studied for 20 rural lakes out of 98 lakes. For surface water, a mean  
275 concentration of 20,264 MP/km<sup>2</sup> with a size class of 333  $\mu$ m - 4.75 mm was observed in  
276 Khövsgöl Lake (Mongolia) (Free et al., 2014). Another study on Khövsgöl Lake showed 0.3  
277 MP/m<sup>3</sup> (Karnaikhov et al., 2020). Nine lakes in the Pampean Region (Argentina) contained

278 0.3 MP/m<sup>3</sup> to 1.9 MP/m<sup>3</sup> with a mean of  $0.9 \pm 0.6$  MP/m<sup>3</sup> and a size class between of 38  $\mu$ m  
279 to 1 mm (Alfonso et al., 2020b). Finally, Lake Sassolo (Switzerland) contained mean  
280 concentrations of  $2.6 \times 10^3$  MP/m<sup>3</sup> and  $4.4 \times 10^3$  fibres/m<sup>3</sup> (natural and synthetic fibres were  
281 excluded from MPs) (Negrete Velasco et al., 2020), 4.1  $\mu$ g of MP fibres/m<sup>2</sup> was reported for  
282 Majcz Lake (Poland) (Kaliszewicz et al., 2020) and, no MP were observed in the surface  
283 water for Dimon Lake (Italy) (**Table S2**). The disparities in the MP concentration between  
284 these lakes could be linked to their characteristics (e.g. volume, surface, and depth). Khövsgöl  
285 Lake (0.3 MP/m<sup>3</sup>) is almost 8 times deeper and 55,200 times larger (on the surface) than  
286 Sassolo Lake ( $2.6 \times 10^3$  MP/m<sup>3</sup>). A voluminous lake exposed to the same MP flux as a  
287 smaller lake will have a lower concentration of MPs. The different sampling methodology  
288 (e.g. net, pump, and mesh size and, surface water or epilimnion) could also explain the  
289 disparities in the MP concentration between lakes. Bottom lake sediments were studied only  
290 in the shallow lake Knudsenheia (Svalbard, Norway) with 400 microparticles of  
291 anthropogenic litter per square meter including 93 MP < 5 mm and 19 MP > 5 mm in each of  
292 the six sediments sampled at the rock surface, thereby leading to an average estimation of  $90$   
293  $\times 10^6$  MP/km<sup>2</sup> (González-Pleiter et al., 2020). In Lake Sassolo (Switzerland), 514 fibres/kg  
294 (dw) and 33 MP/kg (dw) were quantified in the bottom sediment. The MP concentration in  
295 the lakeshore sediments was also variable from  $8 \pm 14 \times 10^6$  to  $563 \pm 1,220 \times 10^6$   
296 particles/km<sup>2</sup> for Siling Co,  $42 \pm 47 \times 10^6$  particles/km<sup>2</sup> for Geren Co,  $117 \pm 126 \times 10^6$   
297 particles/km<sup>2</sup> for Wuru Co, and  $17 \pm 20 \times 10^6$  particles/km<sup>2</sup> for Mujiu Co (all Chinese lakes)  
298 (Zhang et al., 2016). In addition, in Ziway Lake (Ethiopia), the 0-2 cm sediment layer of this  
299 lake contained 33,282 particles/m<sup>3</sup> (from 5,333 particles/m<sup>3</sup> to 97,333 particles/m<sup>3</sup> in the 13  
300 investigated shoreline sites). The MP contamination induced a concentration of a 4 MP/fish.  
301 Benthic and benthopelagic fish species were found to contain a significantly higher number of  
302 plastic particles than planktivorous fish species (Merga et al., 2020). The disparities could be

303 linked to the fish diet and MP concentrations in the lake water and sediment. Depending on  
304 the study, different unit systems, sampling methods, and lake compartments were used,  
305 thereby leading to difficulties in comparing the results. However, all 20 lakes are  
306 contaminated with MPs, and there is a serious need to clearly identify the sources of this  
307 pollution in rural areas.

### 308 3.2. Urban lakes

309 An urban lake is defined as a lake close to at least one town (> 100,000 inhabitants) or one  
310 city (> 1 million inhabitants) (< 50 km). In the scientific literature, MP contamination has  
311 been studied in various compartments of 78 urban lakes out of 98 lakes. Similar to those of  
312 rural lakes, the surface waters of urban lakes are contaminated to various degrees from 10,900  
313 MP/km<sup>2</sup> to 2,090,000 MP/km<sup>2</sup> or from 0.27 MP/m<sup>3</sup> to 34,000 MP/m<sup>3</sup> (**Table S2**). For  
314 instance, concentrations of  $180,900 \pm 229,533$  MP/km<sup>2</sup> with a size class ranging from 112  $\mu$ m  
315 to 5 mm were reported in Qinghai Lake (China), which was mainly due to plastic waste from  
316 tourism (Xiong et al., 2018). Contamination of  $3.1 \times 10^3$  MP/m<sup>3</sup> and  $11.3 \times 10^3$  MP/m<sup>3</sup> with a  
317 size class from 330  $\mu$ m to 5 mm was reported for Ulansuhai Nur Lake (China) (Wang et al.,  
318 2019). In Lake Dziekanowskie and Kalwa Lake (Poland), 4.7  $\mu$ g of MP fibres/m<sup>3</sup> and 4.4  $\mu$ g  
319 of MP fibres/m<sup>3</sup> were reported, respectively (Kaliszewicz et al., 2020). MP pollution in the  
320 surface water of urban lakes showed the lowest concentrations in Zürich (Switzerland) with  
321 10,900.00 MP/km<sup>2</sup> (Faure et al., 2015), Iseo (Italy) with 11,479.00 MP/km<sup>2</sup> (Binelli et al.,  
322 2020), Michigan (USA) with 15,698.00 MP/km<sup>2</sup> (Mason et al., 2016), and Kallavesi (Finland)  
323 with  $0.27 \pm 0.18$  MP/m<sup>3</sup> (Uurasjärvi et al., 2020), Bolsena (Italy) with 2.51 MP/m<sup>3</sup>, Chiusi  
324 (Italy) with 3.02 MP/m<sup>3</sup> (Fischer et al., 2016) and, Simcoe (Canada) with 0.4 – 1.3 MP/m<sup>3</sup>  
325 (Felismino et al., 2020 In Press, Corrected Proof). Therefore, some rural lakes (such as  
326 Khövsgöl with 20,264 MP/km<sup>2</sup> or Sassolo with  $2.6 \times 10^3$  MP/m<sup>3</sup>) seem more contaminated

327 than some urban lakes. The urban Kallavesi Lake (Finland) contains  $0.27 \pm 0.18$  MP/m<sup>3</sup>  
328 (Uurasjärvi et al., 2020), whereas nine rural lakes in Argentina contain  $0.90 \pm 0.60$  MP/m<sup>3</sup>  
329 (Alfonso et al., 2020b). This could be explained by the lake characteristics (depth, surface  
330 area, etc.) that influence the MP concentration. The larger surface area of Kallavesi Lake (473  
331 km<sup>2</sup>), compared with that of Argentinean lakes (mean of 88 km<sup>2</sup>) could be a plausible  
332 explanation for the MP concentration. The different sampling methodologies, with a mesh  
333 size of 333  $\mu$ m in Finland and 38  $\mu$ m in Argentina could also lead to inaccurate result  
334 comparisons. Logically, some urban lakes are more contaminated than rural lakes. For  
335 example, Qinghai Lake (China) showed 9 times more MPs in the surface water than Khövsgöl  
336 Lake (Mongolia) with 180,900 particles/km<sup>2</sup> and 20,264 particles/km<sup>2</sup>, respectively. However,  
337 these disparities could also be linked to the sampling methodology because 112  $\mu$ m and 333  
338  $\mu$ m meshes were used for Qinghai Lake and Khövsgöl Lake, respectively. It is of fundamental  
339 interest to ease the parallel comparison of results by using a consistent methodology to sample  
340 the same size classes of MPs. In addition, quantification by particles/km<sup>2</sup> or particles/m<sup>3</sup> or  
341 use both is of importance. In lake bottom sediments, MP contamination also has different  
342 magnitudes from 0.7 MP/kg to 7,593.0 MP/kg and from 0.8 MP/m<sup>2</sup> to 339.0 MP/m<sup>2</sup>. The  
343 concentrations of lakeshore sediments vary less, from 104 MP/kg to 760 MP/kg. Dongting  
344 Lake (Yueyang, China) contained  $389 \pm 66$  particles/m<sup>3</sup> to  $501 \pm 331$  particles/m<sup>3</sup> in the  
345 lakeshore sediments (Jiang et al., 2018). The MP concentrations in sediments of Lake  
346 Yenagoa (Nigeria) ranged from 1,000 particles/m<sup>3</sup> to 8,330 particles/m<sup>3</sup> during the dry season  
347 and 201 particles/m<sup>3</sup> to 8,370 particles/m<sup>3</sup> during the raining season (Oni et al., 2020). Finally,  
348 the lakeshore sediments of lakes Albano, Bracciano, and Vico (Italy) contained 3.4 g/m<sup>2</sup> of  
349 MPs (Iannilli et al., 2020). The disparities in the MP concentration in sediments could be  
350 linked to the lake depth and polymer density, which modify the fate of these particles in the  
351 water column. For example, Qinghai Lake (China) contained 180,900 particles/km<sup>2</sup> in the

352 surface water and 50 particles/km<sup>2</sup> to 1,290 × 10<sup>6</sup> particles/km<sup>2</sup> in lakeshore sediments,  
353 thereby underlying the role of the sediment (at least temporarily) as a sink. Future studies  
354 should measure the MP sedimentation flux in the water column to better understand the  
355 factors influencing MP accumulation in sediment. MP pollution of beach sediments was  
356 studied for some urban lakes (but not for rural lakes). Various methodologies have also been  
357 used with different unit systems (particles/m<sup>3</sup>, particles/m<sup>2</sup>, particles/g, and g/m<sup>2</sup>). Several  
358 authors have also worked on the MP contamination in fauna. Three fishes out of forty and  
359 eight birds out of nine contained plastic particles in their gut contents in several European and  
360 Swiss lakes (Faure et al., 2015; Faure and De Alencastro, 2016). Amphipods (*Cryptorchestia*  
361 *garbinii*) contained a mean of 2.0 particles/individual and 4.8 particles/individual in the  
362 Italian Lake Albano and Lake Bracciano, respectively (Iannilli et al., 2020). Yuan et al.  
363 (2019) measured 0 particles/fish to 18 particles/fish in Poyang Lake (China). Twenty percent  
364 of the collected fish in Lake Victoria (Uganda) contained MPs (Biginagwa et al., 2016). In the  
365 Lake Mead National Recreation Area (USA), the MP concentration in shellfish ranged from  
366 2.7 particles/organism to 105.0 particles/organism, and the MP concentration in fish ranged  
367 from 0.0 particles/organism to 19.0 particles/organism (Baldwin et al., 2020). Xiong et al.  
368 (2018) detected 5.4 ± 3.6 particles/fish in Qinghai Lake (China). Aquatic fauna is affected by  
369 MP pollution of lakes. However more studies are urgently needed in rural lakes to determine  
370 the order of magnitude related to the different sources. Further studies should focus on MP  
371 contamination in pelagic and benthic fauna to better understand the transfer pathways and  
372 potential impacts on the ecosystems. Finally, snow and ice samples near the shore of a  
373 lacustrine environment have been studied in Finland at Vesijärvi Lake (Scopetani et al.,  
374 2019). The MP concentrations were 117 ± 18 × 10<sup>3</sup> particles/m<sup>3</sup> melted snow and 8 ± 1 × 10<sup>3</sup>  
375 particles/m<sup>3</sup> melted ice. The high porosity of the snow matrix could play an important role in  
376 trapping airborne synthetic fibres (Scopetani et al., 2019). Microplastics contamination in

377 lakes depends on many sources and the functioning of the hydrosystem. It is essential to  
378 conduct studies to inventory all the MP sources and methods of transfer in a specific lake  
379 ecosystem, including its watershed, to understand the mechanisms of MP contamination and  
380 to deduce actions to preserve and/or restore the lacustrine environment. The impact and  
381 transfer of MPs in lake ecosystems are also dependent on the particle size and polymer  
382 composition. Therefore, it is important to identify the main types of MPs in lakes.

### 383 3.3. Predominant types of particles in lakes

#### 384 3.3.1. Microplastic size classes

385 Most of the plastic particles were < 2 mm in all the studied lakes, except for lakes within the  
386 Siling Co basin (northern Tibet), where most particles were between 1 mm and 5 mm (Zhang  
387 et al., 2016). In most cases, the particles were < 1 mm in size (Alfonso et al., 2020a, 2020b;  
388 Boucher et al., 2019; Cable et al., 2017; Di and Wang, 2018; Egessa et al., 2020; Eriksen et  
389 al., 2013; Fischer et al., 2016; Free et al., 2014; González-Pleiter et al., 2020; Irfan et al.,  
390 2020; Mason et al., 2020, 2016; Su et al., 2016; Uurasjärvi et al., 2020), and even < 500  $\mu\text{m}$   
391 (Boucher et al., 2019; Fischer et al., 2016; González-Pleiter et al., 2020; Jiang et al., 2018;  
392 Uurasjärvi et al., 2020; Xiong et al., 2018; Yin et al., 2020; Yuan et al., 2019) (**Table 1**). In  
393 the Great Lakes (Canada, and USA), the number of small plastic particles (106  $\mu\text{m}$ –1.00 mm)  
394 was 130 times higher than that of large particles (1.00–4.75 mm) (Cable et al., 2017). In  
395 Kallavesi Lake (Finland), the particles were 20  $\mu\text{m}$  to 300  $\mu\text{m}$  in size (Uurasjärvi et al., 2020).  
396 Plastic particles on the rock surfaces from the bottom (0.75 m) of a shallow lake in Svalbard  
397 (Norway) were divided as follows: 25 % < 350  $\mu\text{m}$ , 50 % < 670  $\mu\text{m}$ , and 75 % < 1,400  $\mu\text{m}$   
398 (González-Pleiter et al., 2020). Many parameters could affect the size distribution of MPs in  
399 lakes. In Poyang Lake (China), plastic particles in fish are larger (82.1 % > 500  $\mu\text{m}$ ) than  
400 those in surface water and sediments (73.1 % and 57.1% < 500  $\mu\text{m}$ , respectively), thereby

401 suggesting a selective effect during the intake (Yuan et al., 2019). Shoreline sediments in  
402 Lake Victoria (Uganda) contained more MPs of all sizes than in the lake sediments (at a  
403 distance of 250 m and 500 m from the shoreline). This could be due to the proximity of the  
404 shoreline sediments to land-based sources (Egessa et al., 2020). In Lake Yenagoa (Nigeria),  
405 74.9 % of the particles in water and sediment are 1.00–3.00 mm in the dry season and 89.1 %  
406 of the particles are 0.51–1.00 mm in the rainy season (smaller MPs may originate from deep  
407 fragmentation of large plastic debris) (Oni et al., 2020). Various sizes of MP particles could  
408 also be caused by the significant variation in the sampling methodology. For instance, surface  
409 waters are sampled with nets of different mesh sizes, namely 38, 47, 112, 300, 330, 333, and  
410 355  $\mu\text{m}$  or pumped through different filters: 20, 48, 50, 100, 300  $\mu\text{m}$ . Further studies must be  
411 conducted using a unique sampling methodology, unit system (particles/ $\text{km}^2$  or particles/ $\text{m}^3$ ),  
412 and quantification of the different size classes (percentage of the total). Recommendations for  
413 scientific community are provided in Section 1.

### 414 3.3.2. Polymer types

415 The most common types of polymers in lakes (surface water and sediment) were PE and PP  
416 (**Fig. 3**). These results can be linked to the 2018 global demand for polymer, with 17.5 %  
417 LDPE, 12.2 % HDPE and, 19.3 % PP (PlasticsEurope, 2019). The other polymeric materials  
418 produced industrially at the international scale are polyvinyl chloride (PVC) (10 %),  
419 polyurethane (PUR) (7.9 %), polyethylene terephthalate (PET) (7.7 %), polystyrene (PS) and  
420 expanded polystyrene (EPS) (6.4 %) and others (19 %). The variation of MP composition  
421 (polymer types) between lacustrine compartments (surface water, sediment, snow/ice) could  
422 be due to differences in the polymer densities, MP forms, potential MP sources, watershed  
423 characteristics. Overall, PE and PP were predominant, other polymers (e.g.: PVC, PA, PET,  
424 etc.) were found in several lakes but to a lesser extent. The MP composition in the surface  
425 water was similar in most lakes however in some areas, more specific signatures were

426 reported. Those particular signature were observed for one specific scientific paper or one  
427 specific country suggesting a site-specific contamination. Despite the studies number, few  
428 data were available in all lakes, even less for snow and ice. In this last compartments, the  
429 main polymer types were CE and, PA that were mostly found in the form of fibres which was  
430 consistent with an atmospheric origin. As for the particle size, the chemical compositions of  
431 the MPs varied greatly among the lakes. In Lake Yenagoa (Nigeria), the polymer types of the  
432 sampled MPs were related to the season. In surface water and sediment, 72.63 % PET and  
433 10.90 % PVC were observed during the dry season compared with 81.50 % PVC and 4.20 %  
434 LDPE during the rainy season. The high presence of PVC during the rainy season was  
435 thought to originate from the erosion of water pipes of neighbouring communities. The  
436 significant presence of PET during the dry season could be attributed to the importation of  
437 raw material from other regions by three plastic companies producing PET bottles for soft  
438 drinks and water, which was located a few kilometres from the lake shore (Oni et al., 2020).  
439 The release of plastic in Lake Geneva (Switzerland) was estimated to have four main  
440 contributors, namely tire abrasion, the construction sector, the textile industry, and the  
441 packaging industry (Boucher et al., 2019). Tire wear particles (TWPs) are a recent research  
442 topic that requires further investigation to determine its importance in the MP contamination  
443 of lakes. All MPs are first composed of polymers, but they also contain two components,  
444 namely chemical additives (plasticiser, UV blocker, antioxidative compound, flame retardant,  
445 etc.) and chemicals (polycyclic aromatic compounds, trace metallic elements, etc.) absorbed  
446 from the surrounding environment. The leaching of these additives and pollutants from the  
447 plastic surface to the surrounding waters can affect the ecosystem (Wong et al., 2020). In  
448 addition to their chemical composition, the shape of MPs can also greatly determine their fate  
449 and impacts.

### 450 3.3.3. Particle shapes

451 The most commonly identified polymer forms in lakes (surface water and sediment) were  
452 fibres, and, fragments (**Fig. 4**). The MP fibres contamination in lakes could be linked to urban  
453 sources (wastewaters, washing machines). Indeed, urban network was supposed to be a major  
454 MP source in lakes. Investigating MP fibres proportions in relation to watershed inhabitants  
455 could be of high interest, confirming the necessity of watershed characteristics reporting in  
456 future studies (Section 1). Pellets and films were also frequently found compared to foams.  
457 Given the results, not link could be established between the polymer forms and lacustrine  
458 compartments (surface water and sediment). Concerning fishes, only two polymer forms were  
459 observed namely, fibres and fragments. However, very few studies investigated MP pollution  
460 in lake fishes. Moreover, the probability of finding fibre and/or fragment in fish was highest  
461 than other polymer forms given the distribution of the particle forms in lake sediments and  
462 surface waters. Even if the results in this figure were not conclusive, it allowed to present an  
463 overview of polymer forms in all the investigated compartments of the lacustrine  
464 environments across the globe. This lack of data highlighted the importance of increase  
465 studies on freshwater fishes. In Switzerland and Italy, the most represented forms are  
466 fragments and films. However worldwide, fragments and fibres are strongly represented. In  
467 nine lakes in the Pampean Region (Argentina), MP fibres ranged from 66.70 % to 96.40 %  
468 (Alfonso et al., 2020b). Determining the particle shapes is essential to understanding the fate  
469 of MPs in lake ecosystems (e.g. fibres and foams will probably float) and the impacts on biota  
470 (e.g. coarse fragments may injure the digestive system in fish while nanoparticles may  
471 translocate into organs).

## 472 4. Sources of microplastics in lakes

### 473 4.1. Introduction

474 The sources of MPs in lakes are numerous, and MPs can be transferred from various origins.  
475 Plastic pollution has three origins, namely macroplastic fragmentation in the environment,  
476 wear and tear of products that contain plastic polymers (e.g. fragmentation of textile fibres  
477 during washing, paints, and tires), and intentional origin (e.g. exfoliants, industrial abrasions,  
478 and drug vectorisation) (INERIS, 2018). The principal top-down MP sources are tires, paint  
479 markings, plastic resins, and clothing made of synthetic fibres (Hann et al., 2018). Moreover,  
480 MP pollution is intensified by climatic conditions. According to previous research, there are  
481 several climate change impacts concerning MPs in aquatic ecosystems (Welden and Lusher,  
482 2017; Zhang Yafei et al., 2020). MP emissions to the environment and transport increase  
483 through precipitation, flooding, glacier melting, and runoff; plastic in the atmosphere  
484 intensifies with the increase in strong wind events; MP suspension from sediments increases  
485 through wind and water currents as well as temperature changes in water; and plastic retention  
486 increases through higher evaporation in lakes. (Boucher et al., 2019b) assumed “the potential  
487 role of big lakes as a barrier preventing more plastic from reaching the oceans”. The previous  
488 works point out the lack of integrated studies considering the whole ecosystem, which is  
489 crucial for future investigations of MP pollution in lakes integrated at the catchment scale.  
490 This should help to understand the involvement of each potential MP contamination source in  
491 lakes, namely soil leaching and road runoff, wastewater treatment plant (WWTP) discharge,  
492 atmospheric fallout, and in-situ macroplastic fragmentation. Indeed, these sources are  
493 influenced by the population density (e.g. increase in single-use plastics), road density,  
494 impermeable watershed area, industrial activity of plastic production/transformation, climatic  
495 conditions, and lake characteristics. Public authorities require these elements to act on major  
496 sources and implement new policies to reduce the plastic contamination in lakes.

## 497 4.2. Macroplastic fragmentation

498 Fragmentation of macroplastic waste could be one of the main sources of MPs in lakes. Most  
499 of the knowledge on plastic fragmentation is based on in vitro experiments with ultraviolet  
500 (UV), thermal, or bio-degradation. Limited knowledge was acquired under real environmental  
501 conditions in which degradation mechanisms occur simultaneously or successively.

502 Environmental conditions (UV irradiation intensity, pH, and temperature) and chemical and  
503 physical properties of plastic polymers (density, crystallinity, and the presence of additives)  
504 play crucial roles in plastic fragmentation. Plastic polymers from oligotrophic lakes can break  
505 up more easily with high UV penetration and weak biofouling, which reinforce the polymer  
506 durability (Andrady, 2011, 2003; Free et al., 2014). In lakes at high altitudes, low  
507 temperatures and snow cover can better maintain the plastic particle integrity (Andrady, 2003;  
508 Free et al., 2014). However, macroplastic degradation is weaker in freshwater than in marine  
509 ecosystems owing to weaker current and limited wave action (Vaughan et al., 2017).

510 However, littoral waves, especially in vast lakes, can cause higher fragmentation rates (Imhof  
511 et al., 2013). Low turbulence and wave action in some freshwater ecosystems lead to longer  
512 pollution persistence. The estimations of plastic longevity range from hundreds to thousands  
513 of years (Barnes et al., 2009). In 2016, plastic particles were found in each sediment sample  
514 from Lake Geneva (Switzerland), which mainly originated from macroplastic fragmentation  
515 (Faure and De Alencastro, 2016). Plastic pollution in Lake Michigan (USA) also revealed a  
516 high concentration of secondary MPs (from the breakdown of larger plastics), especially  
517 fragments (Mason et al., 2016). Near Dongting Lake (Yueyang, China) and Lake Victoria  
518 (Uganda), synthetic fishing nets have a strong impact on plastic contamination and constitute  
519 a major source of secondary MPs (Egessa et al., 2020; Yin et al., 2020). The predominance of  
520 coloured MP particles could reflect macroplastic fragmentation (Migwi et al., 2020). Finally,  
521 a large amount of cellulose acetate, which is one of the main cigarette filter components, was  
522 reported in Lake Ontario (Canada/USA) (Grbić et al., 2020). The prevalence of small plastic

523 particles from fragmentation was also reported in Khövsgöl Lake (Mongolia) (Free et al.,  
524 2014) and Taigu and Poyang lakes (China) (Su et al., 2016; Yuan et al., 2019). Plastic  
525 degradation processes, namely cracking, splitting, and biofilm formation, seem globally  
526 similar in freshwater and marine ecosystems (Imhof et al., 2013; Li et al., 2020; Zbyszewski  
527 and Corcoran, 2011). However, little is known about the kinetics of fragmentation in lakes.  
528 After 25 weeks of accelerated weathering in water under xenon lamp irradiation, low-density  
529 polyethylene (LDPE) films split into smaller fragments (62 % on average) corresponding to  
530 the definition of MPs (< 5 mm), thereby suggesting that long residence times in water lead to  
531 MP fragmentation and contamination of the aquatic environment (Julienne et al., 2019).  
532 Mechanical properties at the polymer surface play a major role in the fragmentation pathway  
533 whereas the fabrication process may influence the propagation direction of the cracks. The  
534 size distribution of plastic fragments may be linked not only to the nature of the polymer, but  
535 also to its manufacturing process (Julienne et al., 2019). The number of polymer particles  
536 fragmented by UV exposure and mechanical abrasion increased with decreasing size in all  
537 polymer types. According to Song et al. (2017), mechanical abrasion by sand minimally  
538 fragmented polyethylene (PE) and polypropylene (PP) without UV photo-oxidation.  
539 Moreover, the rate of fragmentation increased with UV exposure depending on the polymer  
540 type. Twelve months of UV exposure and two months of mechanical abrasion of PP and PE  
541 produced  $6,084.0 \pm 1,061.0$  particles/pellet and  $20.0 \pm 8.3$  particles/pellet (mean  $\pm$  SD),  
542 respectively. To roughly quantify the fragmentation flux in the lake, we considered a mean  
543 fragmentation rate of 3,052 MP/pellet (independent of the polymeric composition) and the  
544 average weight of one pellet (Song et al., 2017) to 20 mg. As the concentration of  
545 macroplastics in the surface water of Lake Geneva was measured as 44,000 mg/km<sup>2</sup> (Grand  
546 Lac) and 27,000 mg/km<sup>2</sup> (Petit Lac) (Faure et al., 2015) we estimated a fragmentation flux  
547 from the surface as high as  $1.1 \times 10^8$  MP/y (**SI Calculations**). We also estimated the

548 macroplastics concentration of surface waters in worldwide lakes with available data in  
549 literature by calculating the mean MP concentration (mp/km<sup>2</sup>) in surface waters of lakes,  
550 namely 2,808.98 macroplastics/km<sup>2</sup> (based on Cable et al., 2017; Faure et al., 2015; Faure et  
551 al., 2012; Mason et al., 2016). We linked this macroplastics concentration estimate to MP  
552 concentration in the corresponding lakes surface waters (based on Cable et al., 2017; Faure et  
553 al., 2015; Faure et al., 2012; Mason et al., 2016), namely 133,741 mp/km<sup>2</sup>. Additional studies  
554 are however required to clearly comprehend the mechanisms, quantify the kinetics of  
555 fragmentation within the environment, and determine the precise origins of macroplastics  
556 waste from industrial, agricultural, and domestic use, among others.

### 557 4.3. Wastewater

558 WWTPs might be an important source of MPs in lacustrine environments depending on the  
559 WWTP setup, population density, or discharge of wastewater into lakes (Bretas Alvim et al.,  
560 2020). Wastewater contains both types of MPs, namely primary (tiny particles designed for  
561 commercial use, e.g. personal care products) and secondary (from the breakdown of larger  
562 plastics, e.g. textile fibres in domestic wastewater or tire particles in stormwater). For  
563 instance, the influent of a Hong Kong WWTP contained  $7.1 \pm 6.0$  particles/L to  $12.8 \pm 5.8$   
564 particles/L with intra-day variations of  $6.6 \times 10^8$  particles/d and  $1.2 \times 10^9$  particles/d (Cao et  
565 al., 2020). Several experiments confirmed that textile fibres are an important component of  
566 MPs in wastewater. One item of clothing produces more than 1,900 fibres per wash (Browne  
567 et al., 2011); one fleece produces approximately 110,000 fibres (Carney Almroth et al., 2018),  
568 and 5 kg of polyester (PES) cloth can release approximately 6,000,000 microfibrils (De Falco  
569 et al., 2019). Fibres represent 70 % of MPs in tributaries from WWTPs (Talvitie et al., 2016).  
570 The efficiency of MP removal in WWTPs varies between 88.0 and 99.9 % (Uddin et al.,  
571 2020). The average concentration of MPs and other anthropogenic particles (with a Fourier-

572 Transform Infrared spectroscopy or Raman spectroscopy signal of an anthropogenic dye)  
573 found in wastewater effluent was  $13.3 \pm 15.5$  particles/L near Lake Ontario (USA and  
574 Canada). Fibres were also the dominant category in WWTP effluent with 90 % relative  
575 abundance (Grbić et al., 2020). The disposal of treated effluents can add approximately  $1.5 \times$   
576  $10^{15}$  MP/y to the aquatic environment, whereas the discharge of untreated effluent is likely to  
577 add  $3.9 \times 10^{16}$  MP/y at the global scale (Uddin et al., 2020).

#### 578 4.4. Stormwater, agricultural runoff, and tributaries

579 Surface runoff and lake tributaries could also be significant sources of MPs according to the  
580 lake catchment basin characteristics and climatic conditions. Rivers represent an important  
581 source of MPs because they can collect WWTP effluents, untreated stormwater, road runoff,  
582 and agricultural runoff. As an example, the MP abundance increased in the Cooks River  
583 estuary (Sydney, Australia) from 400 particles/m<sup>3</sup> to 17,383 particles/m<sup>3</sup> before 2 days of  
584 heavy rain events (Hitchcock, 2020). Rain and storm events may be key moments of MP  
585 contamination in lake ecosystems. In Lake Ontario (USA and Canada), MPs in sediment  
586 samples were mostly carried by nearby tributaries (Ballent et al., 2016). The average  
587 concentrations of anthropogenic particles were  $15.4 \pm 7.9$  particles/L in stormwater runoff  
588 (sampling in urban rivers during wet weather) and  $0.9 \pm 1.3$  particles/L in agricultural runoff  
589 near Lake Ontario (USA and Canada) (Grbić et al., 2020). The most abundant particles in  
590 stormwater samples were microfibrils (41 %) and tire and road wear particles (TRWPs) (22  
591 %) (Grbić et al., 2020). Sewer overflow systems and urban runoff are also major sources of  
592 MPs in different subalpine lakes (Faure and De Alencastro, 2016). For example, the plastic  
593 inflow in Lake Geneva (Switzerland) was estimated to be between 34 t/y and 83 t/y (average  
594 59 t/y) with strong inputs from road and urban runoff after precipitation (Boucher et al.,  
595 2019). The mass of TWPs ultimately entering the aquatic environment strongly depends on

596 the extent of collection and treatment of road runoff, which is highly variable (Wagner et al.,  
597 2018). Up to 11,000 t/y of TWP from German highways can enter surface waters (Wagner et  
598 al., 2018). Agricultural runoff can also contaminate tributaries and the lacustrine environment  
599 because of plastic mulching, WWTP sludge spreading or even irrigation with recycled  
600 wastewater (an approach used in a few countries). Plastic mulching is an agricultural  
601 technique used to reduce evaporation, suppress weeds, conserve moisture and increase soil  
602 temperature. In China, MP concentrations in agricultural soil increased over time with 5, 15,  
603 and 24 y of continuous mulching with  $80.3 \pm 49.3$ ,  $308.0 \pm 138.1$ , and  $1,075.6 \pm 346.8$   
604 particles/kg soil, respectively (Huang et al., 2020). The spreading of liquid or dewatered  
605 WWTP sludge may also be a source of MP to lakes through agricultural soil leaching. There  
606 is still uncertainty about how the plastic debris interacts with the microorganisms constituting  
607 the sludge during biological wastewater treatment processes. However, the MP concentrations  
608 varied between 8.2 particles/g (dw) and 301.4 particles/g (dw) in sludge samples collected  
609 from a municipal WWTP based on the conventional activated sludge process in Finland  
610 (Lares et al., 2018). In addition, play areas and sports fields can also generate large quantities  
611 of MPs in stormwater systems: they are considered the fifth major source of MPs in Germany  
612 (Bertling et al., 2018).

#### 613 4.5. Atmospheric fallout

614 Atmospheric fallout enriched in MPs has been observed in urban, industrial, and remote areas  
615 (Zhang et al., 2016, 2019; Yulan Zhang et al., 2020). Atmospheric transport constitutes a  
616 large flux of plastic particles in the environment, but it has been poorly studied (Horton and  
617 Dixon, 2017). In the Paris Metropolitan Area,  $3.5 \times 10^{10}$  and  $7.6 \times 10^{10}$  synthetic fibres could  
618 originate from the atmosphere per year, thereby leading to an atmospheric flux of  $18 \times 10^6$   
619 synthetic fibres/(km<sup>2</sup>·d) to  $37 \times 10^6$  synthetic fibres/(km<sup>2</sup>·d) (Rachid Dris, 2016). Intense

620 precipitation (rain, storm and snow) probably increases MP atmospheric fallout via washout  
621 of MPs (Allen et al., 2019; Dris et al., 2016a, 2016b). In the absence of wastewater discharge  
622 to the Hampstead No. 1 Lake near London, atmospheric fallout was considered the main  
623 source of MPs as fibres were mostly observed in sediment samples (Turner et al., 2019).  
624 Synthetic textiles and TRWPs are considered the major sources of MPs in the atmosphere  
625 with a majority of fibres and fragments (Yulan Zhang et al., 2020). To the best of our  
626 knowledge, the mean fluxes of MPs in atmospheric fallout vary from  $36 \times 10^6$   
627 particles/(km<sup>2</sup>·d) to  $770 \times 10^6$  particles/(km<sup>2</sup>·d) (Allen et al., 2019; Cai et al., 2017; Dris et al.,  
628 2016a, 2015; Klein and Fischer, 2019; Wright et al., 2020; Zhou et al., 2017) (**Fig. 5**).  
629 Therefore, the mean flux of MPs in atmospheric fallout can be estimated as  $287 \times 10^6$   
630 particles/(km<sup>2</sup>·d). Even if data from atmospheric fallout are more detailed than data from  
631 other MP sources to lakes worldwide, further studies should imperatively determine both the  
632 atmospheric MP flux and MP concentration in surface water for one specific lake area to  
633 understand the real impact of atmospheric fallout. Future work is urgently needed to clearly  
634 quantify the MP flux to lake water at the catchment basin scale, namely in-situ macroplastics  
635 fragmentation, wastewater or stormwater discharge, agricultural soil runoff, and atmospheric  
636 fallout.

#### 637 4.6. Estimates of source contributions to lake contamination by microplastics

638 Depending on the catchment features, the relative contribution of each source of MPs to a  
639 lake may considerably differ both in nature and intensity. As a recent example, Lake Ontario  
640 (USA and Canada) is exposed to various sources of MPs. Some of them have been quantified  
641 with MP (> 125 µm) concentrations (Grbić et al., 2020), namely WWTP discharge,  
642 impermeable urban surface runoff, and agricultural runoff. The concentrations are highly  
643 variable with  $15,400 \pm 7,900$  particles/m<sup>3</sup> in stormwater runoff (relative abundances of fibres

644 of 41 %, and TRWPs of 22 %),  $13,300 \pm 15,500$  particles/m<sup>3</sup> in wastewater effluent (90 %  
645 fibres), and  $900 \pm 1,300$  particles/m<sup>3</sup> in agricultural runoff (no dominant category).  
646 Unfortunately, the intensity of these sources (as a flux of MPs to the lake) cannot be estimated  
647 as the flow rates have not been measured. Boucher et al. (2019) estimated, for the first time to  
648 our best knowledge, the contributions of the different MP sources to a lake contamination.  
649 The approach based on field measurement showed the following MP (> 300 µm to 5 mm)  
650 contributions: 41.3 t/y river discharge (wet period), 8.9 t/y urban runoff, 5.6 t/y river  
651 discharge (dry period), 2.6 t/y storm overflow, 0.05 t/y waste water effluent and, 0.03 t/y  
652 atmospheric fallout. In Western Europe waste management systems are efficient (in Germany  
653 MP retention of 97% (Mintenig et al., 2017)), these estimated fluxes could be higher in  
654 countries with less efficient waste management systems. However, the watershed  
655 characteristics are essential to understand the MP contamination in a specific lake. In our  
656 review, we estimated the MP contribution of macroplastics fragmentation to reach  $1.1 \times 10^8$   
657 MP/y corresponding to approximately 0.03 t/y which is similar the atmospheric fallouts  
658 estimated by Boucher et al. (2019b) (0.03 t/y > 300 µm). Our own estimations on fallout flux  
659 was 0.2 t/y (MPs > 300 µm) (**SI Calculations**) which was of the same order of magnitude.  
660 However, these calculations were strongly dependent on size particle, volume particle and,  
661 density. Therefore, reporting MP size distributions, watershed characteristics, polymer types  
662 and forms are essential in future studies and to improve estimations (Section 1). Macroplastics  
663 fragmentation and atmospheric fallouts could be weak MP sources in lakes. The main  
664 limitation on atmospheric deposition fluxes is the lack of data on rural areas. Consequently,  
665 the main sources of MP to lacustrine environment would be the leaching of MP from rural  
666 areas or other sources such as road runoff which are not treated by WWTP. Given these  
667 results it would be important for future studies to study the MP flux from urban and  
668 impermeabilized watershed which are a significant source of MPs in the environment and, on

669 atmospheric fallout which could be the main source of MP in more remote lakes. We also  
670 suggest to investigate tributaries, from now on it is absolutely necessary to work on potential  
671 MP sources linked to the investigated lake as Hu et al. (2020) wrote “rivers became an  
672 important source of MPs in lakes and studying surrounding rivers while working on a lake is  
673 essential”. The scientific community is aware of MP in freshwaters, now it is time to  
674 understand where they come from and the MP quantities of each potential sources. Future  
675 studies should focus on the different point sources and diffuse sources in contrasting  
676 watersheds (urbanised or rural) to precisely quantify the intensity and composition of each  
677 source. This will help public authorities, local policies, or companies to guide best  
678 management practices and future mitigation strategies for the reduction in MP pollution,  
679 which is highly variable from one lake to another.

## 680 5. Conclusion

681 MP pollution was reviewed in 98 lakes worldwide with a distinction between rural (20) and  
682 urban (78) lakes, according to the supposed main influence of the nearby population density.  
683 Remote lakes should be also studied to clearly quantify the fluxes of various MP sources. In  
684 this review, we noticed MP pollution in each of the 98 investigated lakes. Some lakes showed  
685 MP contamination in the surface water, sediments (lake, shoreline, and beach), tributaries,  
686 aquatic fauna, ice and/or snow. The common MP size class studied was 300  $\mu\text{m}$  to 1 mm. The  
687 MP concentrations were sometimes higher in rural lakes than in urban lakes. We insist on the  
688 importance of identifying MPs sources (macroplastics fragmentation, wastewater or  
689 stormwater discharge, agricultural and road runoff, tributaries, and atmospheric fallout) and  
690 quantifying their respective flux to the lake to inform public authorities in order to take  
691 accurate actions for plastic pollution reduction. For example, input plastic flows in Lake  
692 Geneva from land were estimated to be 55 t/y; the plastic stocks were 0.10 t on the surface,

693 10.00 t on the banks, and 580.00 t in the sediments, and the plastic out flows were 5 t/y in the  
694 outflow of the Rhone river, 10 t/y in coastal clean-ups, and 40 t/y in sediments (Boucher et al.,  
695 2019). This study will allow public authorities to act on precise areas and set effective  
696 systems, such as stormwater retention tanks to avoid WWTP bypass in tributaries, tertiary  
697 treatment in WWTPs to selectively remove plastic fibres, stormwater retention ponds to  
698 intercept tire particles from road runoff, mainstream awareness campaigns, intensification of  
699 the number of recycling and garbage bins (in urban but also rural areas), campaigns to raise  
700 public awareness concerning remote pollution, bans on single-use plastic, use of reusable  
701 packing and dishes in every sector; construction of new recycling centres and penalties for  
702 non-reusable packaging.

703 References :

- 704 Alfonso et al., 2020a M.B. Alfonso, A.H. Arias, M.C. Piccolo  
705 **Microplastics integrating the zooplanktonic fraction in a saline lake of Argentina:**  
706 **influence of water management**  
707 Environ. Monit. Assess., 192 (2020), pp. 117, 10.1007/s10661-020-8080-1  
708
- 709 Alfonso et al., 2020b M.B. Alfonso, F. Scordo, C. Seitz, G.M.M. Manstretta, A.C. Ronda,  
710 A.H. Arias, J.P. Tomba, L.I. Silva, G.M.E. Perillo, M.C. Piccolo  
711 **First evidence of microplastics in nine lakes across Patagonia (South America)**  
712 Sci. Total Environ., 733 (2020), pp. 139385, 10.1016/j.scitotenv.2020.139385  
713
- 714 Allen et al., 2019 S. Allen, D. Allen, V.R. Phoenix, G. Le Roux, P. Durántez Jiménez, A.  
715 Simonneau, S Binet, D. Galop  
716 **Atmospheric transport and deposition of microplastics in a remote mountain**  
717 **catchment**  
718 Nat. Geosci., 12 (2019), pp. 339–344, 10.1038/s41561-019-0335-5  
719
- 720 Anderson et al., 2017 P.J. Anderson, S. Warrack, V. Langen, J.K. Challis, M.L. Hanson, M.D.  
721 Rennie  
722 **Microplastic contamination in Lake Winnipeg, Canada**  
723 Environ. Pollut., 225 (2017), pp. 223–231, 10.1016/j.envpol.2017.02.072  
724
- 725 Andrady, 2011 A. L. Andrady  
726 **Microplastics in the marine environment**  
727 Mar. Pollut. Bull., 62 (2011), pp. 1596–1605, 10.1016/j.marpolbul.2011.05.030  
728
- 729 Andrady, 2003 A.L. Andrady  
730 **Plastics and the environment**  
731 John Wiley & Sons publishing, pp. 1-747  
732
- 733 Arias-Andres et al., 2018 M. Arias-Andres, U. Klümper, K. Rojas-Jimenez, H.-P. Grossart  
734 **Microplastic pollution increases gene exchange in aquatic ecosystems**  
735 Environ. Pollut., 237 (2018), pp. 253–261, 10.1016/j.envpol.2018.02.058  
736
- 737 Arthur et al., 2009 C. Arthur, J. Baker, H. Bamford  
738 **Proceedings of the international research workshop on the occurrence, effects,**  
739 **and fate of microplastic marine debris**  
740 NOAA, Technical Memorandum  
741
- 742 Baldwin et al., 2020 A.K. Baldwin, A.R. Spanjer, M.R. Rosen, T. Thom  
743 **Microplastics in Lake Mead National Recreation Area, USA: occurrence and**  
744 **biological uptake**  
745 PLOS ONE, 15 (2020), pp. e0228896, 10.1371/journal.pone.0228896  
746
- 747 Ballent et al., 2016 A. Ballent, P.L. Corcoran, O. Madden, P.A. Helm, F.J. Longstaffe  
748 **Sources and sinks of microplastics in Canadian Lake Ontario nearshore,**  
749 **tributary and beach sediments**  
750 Mar. Pollut. Bull., 110 (2016), pp. 383–395, 10.1016/j.marpolbul.2016.06.037

- 751 Barnes et al., 2009 D.K.A Barnes, F. Galgani, R.C. Thompson, M. Barlaz  
752 **Accumulation and fragmentation of plastic debris in global environments**  
753 Phil. Trans. R. Soc. B, 364 (2009), pp. 1985–1998 10.1098/rstb.2008.0205  
754
- 755 Bellasi et al., 2020 A. Bellasi, G. Binda, A. Pozzi, S. Galafassi, P. Volta, R. Bettinetti  
756 **Microplastic contamination in freshwater environments: a review, focusing on**  
757 **interactions with sediments and benthic organisms**  
758 Environments, 7 (2020), pp. 30, 10.3390/environments7040030  
759
- 760 Bertling et al., 2018 J. Bertling, R. Bertling, L. Hamman  
761 **Kunststoffe in der Umwelt: Mikro- und Makroplastik. Ursachen, Mengen,**  
762 **Umweltschicksale, Wirkungen, Lösungsansätzen, Empfehlungen**  
763 Fraunhofer-Institut für Umwelt-, Sicherheits- und Energietechnik UMSICHT.  
764 10.24406/umsicht-n-497117  
765
- 766 Bertoldi et al., 2020 C. Bertoldi, L.Z. Lara, F.A. de L. Mizushima, F.C.G. Martins, M.A.  
767 Battisti, R. Hinrichs, A.N. Fernandes  
768 **First evidence of microplastic contamination in the freshwater of Lake Guaíba,**  
769 **Porto Alegre, Brazil**  
770 Sci. Total Envir., 759 (2020), pp. 143503  
771
- 772 Bharath et al., 2021 K.M. Bharath, S. Srinivasalu, U. Natesan, R. Ayyamperumal, S.N.  
773 Kalam, S. Anbalagan, K. Sujatha, C. Alagarasan  
774 **Microplastics as an emerging threat to the freshwater ecosystems of Veeranam**  
775 **lake in south India: A multidimensional approach**  
776 Chemosphere, 264 (2021), pp. 128502, 10.1016/j.chemosphere.2020.128502  
777
- 778 Biginagwa et al., 2016 F.J. Biginagwa, B.S. Mayoma, Y. Shashoua, K. Syberg, F.R.  
779 **First evidence of microplastics in the African Great Lakes: recovery from Lake**  
780 **Victoria Nile perch and Nile tilapia**  
781 J. of Great Lakes Res., 42 (2016), pp. 146–149, 10.1016/j.jglr.2015.10.012  
782
- 783 Binelli et al., 2020 A. Binelli, L. Pietrelli, S. Di Vito, L. Coscia, M. Sighicelli, C.D. Torre,  
784 C.C. Parenti, S. Magni, S.  
785 **Hazard evaluation of plastic mixtures from four Italian subalpine great lakes on**  
786 **the basis of laboratory exposures of zebra mussels**  
787 Sci. Total Envir., 699 (2020), pp. 134366, 10.1016/j.scitotenv.2019.134366  
788
- 789 Bordós et al., 2019 G. Bordós, B. Urbányi, A. Micsinai, B. Kriszt, Z. Palotai, I. Szabó, Z.  
790 Hantosi, S. Szoboszlai  
791 **Identification of microplastics in fish ponds and natural freshwater environments**  
792 **of the Carpathian basin, Europe**  
793 Chemosphere, 216 (2019), pp. 110-116, 10.1016/j.chemosphere.2018.10.110  
794
- 795 Boucher et al., 2019 J. Boucher, F. Faure, O. Pompini, Z. Plummer, O. Wieser, L. Felipe de  
796 Alencastro  
797 **(Micro) plastic fluxes and stocks in Lake Geneva basin**  
798 Trends Anal. Chem., 112 (2019), pp. 66–74, 10.1016/j.trac.2018.11.037  
799
- 800 Bretas Alvim et al., 2020 C. Bretas Alvim, J.A. Mendoza-Roca, A. Bes-Piá

801 **Wastewater treatment plant as microplastics release source – Quantification and**  
802 **identification techniques**  
803 J. of Envir. Manag., 255 (2020), pp. 109739, 10.1016/j.jenvman.2019.109739  
804

805 Browne et al., 2011 M.A. Browne, P. Crump, S.J. Niven, E. Teuten, A. Tonkin, T. Galloway,  
806 R. Thompson  
807 **Accumulation of microplastic on shorelines worldwide: sources and sinks**  
808 Environ. Sci. Technol., 45 (2011), pp. 9175–9179, 10.1021/es201811s  
809

810 Cable et al., 2017 R.N. Cable, D. Beletsky, R. Beletsky, K. Wigginton, B.W. Locke, M.B.  
811 Duhaime  
812 **Distribution and modeled transport of plastic pollution in the Great Lakes, the**  
813 **world’s largest freshwater resource**  
814 Front. Environ. Sci., 5 (2017), pp. 45, 10.3389/fenvs.2017.00045  
815

816 Cai et al., 2017 L. Cai, J. Wang, J. Peng, Z. Tan, Z. Zhan, X. Tan, Q. Chen  
817 **Characteristic of microplastics in the atmospheric fallout from Dongguan city,**  
818 **China: preliminary research and first evidence**  
819 Environ. Sci. Pollut. Res., 24 (2017), pp. 24928–24935, 10.1007/s11356-017-0116-x  
820

821 Cao et al., 2020 Y. Cao, Q. Wang, Y. Ruan, R. Wu, L. Chen, K. Zhang, P.K.S. Lam  
822 **Intra-day microplastic variations in wastewater: a case study of a sewage**  
823 **treatment plant in Hong Kong**  
824 Mar. Pollut. Bull., 160 (2020), pp. 111535, 10.1016/j.marpolbul.2020.111535  
825

826 Carney Almroth et al., 2018 B.M. Carney Almroth, L. Åström, S. Roslund, H. Petersson, M.  
827 Johansson, N.-K. Persson  
828 **Quantifying shedding of synthetic fibers from textiles; a source of microplastics**  
829 **released into the environment**  
830 Envir. Sci. Pollut. Res., 25 (2018), pp. 1191–1199, 10.1007/s11356-017-0528-7  
831

832 Carpenter & Smith, 1972 E.J. Carpenter, K.L. Smith  
833 **Plastics on the Sargasso Sea Surface**  
834 Science, 175 (1972), pp. 1240–1241  
835

836 Conkle et al., 2018 J.L. Conkle, C.D. Báez Del Valle, J.W. Turner  
837 **Are we underestimating microplastic contamination in aquatic environments?**  
838 Envir. Manag., 61 (2018), pp. 1–8, 10.1007/s00267-017-0947-8  
839

840 Cox et al., 2019 K.D Cox, G.A. Covernton, H.L. Davies, J.F. Dower, F. Juanes, S.E. Dudas  
841 **Human consumption of microplastics**  
842 Environ. Sci. Technol., 53 (2019), pp. 7068–7074, 10.1021/acs.est.9b01517  
843

844 De Falco et al., 2019 F. De Falco, E. Di Pace, M. Cocca, M. Avella  
845 **The contribution of washing processes of synthetic clothes to microplastic**  
846 **pollution** Sci. Rep., 9 (2019), pp. 6633, 10.1038/s41598-019-43023-x  
847

848 Di & Wang, 2018 M. Di, J. Wang  
849 **Microplastics in surface waters and sediments of the Three Gorges Reservoir,**  
850 **China**

- 851 Sci. Total Envir., 616–617 (2018), pp. 1620–1627, 10.1016/j.scitotenv.2017.10.150  
852
- 853 Dris, 2016 R. Dris  
854 **First assesement of sources and fate of macro and micro plastics in urban**  
855 **hydrosystems: Case of Paris megacity**  
856 Hydrology, Université Paris-Est  
857
- 858 Dris et al., 2016a R. Dris, J. Gasperi, M. Saad, C. Mirande, B. Tassin  
859 **Synthetic fibers in atmospheric fallout: A source of microplastics in the**  
860 **environment?**  
861 Mar. Pollut. Bull., 104 (2016), pp. 290–293, 10.1016/j.marpolbul.2016.01.006  
862
- 863 Dris et al., 2015 R. Dris, H. Imhof, W. Sanchez, J. Gasperi, F. Galgani, B. Tassin, C. Laforsch  
864 **Beyond the ocean: contamination of freshwater ecosystems with (micro-)plastic**  
865 **particles**  
866 Environ. Chem. Lett., 12 (2015), pp. 539-550, 10.1071/EN14172  
867
- 868 Dris et al., 2016b R. Dris, L. Lahens, V. Rocher, J. Gasperi, B. Tassin  
869 **Premières investigations sur les microplastiques en Seine**  
870 PIREN-Seine, phase VII  
871
- 872 Egessa et al., 2020 R. Egessa, A. Nankabirwa, R. Basooma, R. Nabwire  
873 **Occurrence, distribution and size relationships of plastic debris along shores and**  
874 **sediment of northern Lake Victoria**  
875 Environ. Pollut., 257 (2020), pp. 113442, 10.1016/j.envpol.2019.113442  
876
- 877 Eriksen et al., 2013 M. Eriksen, S. Mason, S. Wilson, C. Box, A. Zellers, W. Edwards, H.  
878 Farley, S. Amato  
879 **Microplastic pollution in the surface waters of the Laurentian Great Lakes**  
880 Mar. Pollut. Bull., 77 (2013), pp. 177–182, 10.1016/j.marpolbul.2013.10.007  
881
- 882 Faure et al., 2012 F. Faure, M. Corbaz, H. Baecher, L. Felippe  
883 **Pollution due to plastics and microplastics in Lake Geneva and in the**  
884 **Mediterranean Sea**  
885 Arch. Sci., 65 (2012), pp. 157-164  
886
- 887 Faure & Alencastro, 2016 F. Faure, F. De Alencastro  
888 **Microplastiques : situation dans les eaux de surface en Suisse**  
889 Fachartikel Aqua & Gas, 4 (2016), pp. 72-77  
890
- 891 Faure et al., 2015 F. Faure, C. Demars, O. Wieser, M. Kunz, L.F. de Alencastro  
892 **Plastic pollution in Swiss surface waters: nature and concentrations, interaction**  
893 **with pollutants**  
894 Environ. Chem., 12 (2015), pp. 582-591, 10.1071/EN14218  
895
- 896 Felismino et al., 2020 M.E.L. Felismino, P.A. Helm, C.M. Rochman  
897 **Microplastic and other anthropogenic microparticles in water and sediments of**  
898 **Lake Simcoe**  
899 J. Great Lakes Res., 10.1016/j.jglr.2020.10.007  
900

- 901 Fischer et al., 2016 E.K. Fischer, L. Paglialonga, E. Czech, M. Tamminga  
 902 **Microplastic pollution in lakes and lake shoreline sediments – A case study on**  
 903 **Lake Bolsena and Lake Chiusi (central Italy)**  
 904 Environ. Pollut., 213 (2016), pp. 648–657, 10.1016/j.envpol.2016.03.012  
 905
- 906 C.M. Free, O.P. Jensen, S.A. Mason, M. Eriksen, N.J. Williamson, B. Boldgiv  
 907 **High-levels of microplastic pollution in a large, remote, mountain lake**  
 908 Mar. Pollut. Bull., 85 (2014), pp. 156–163, 10.1016/j.marpolbul.2014.06.001  
 909
- 910 Gasperi et al., 2018 J. Gasperi, S.L. Wright, R. Dris, F. Collard, C. Mandin, M. Guerrouache,  
 911 V. Langlois, F.J. Kelly, B. Tassin  
 912 **Microplastics in air: Are we breathing it in?**  
 913 Curr. Opin. Environ. Sci. Health, 1 (2018), pp. 1–5, 10.1016/j.coesh.2017.10.002  
 914
- 915 Gateuille et al., 2020 D. Gateuille, J. Dusaucy, F. Gillet, J. Gasperi, R. Dris, G. Tourreau, E.  
 916 Naffrechoux  
 917 **Microplastic contamination in remote alpine lakes**  
 918 EGU General Assembly, Vienne, Austria, 10.5194/egusphere-egu2020-1292  
 919
- 920 González-Pleiter et al., 2020 M. González-Pleiter, D. Velázquez, C. Edo, O. Carretero, J.  
 921 Gago, Á. Barón-Sola, L.E. Hernández, I. Yousef, A. Quesada, F. Leganés, R. Rosal, F.  
 922 Fernández-Piñas  
 923 **Fibers spreading worldwide: Microplastics and other anthropogenic litter in an**  
 924 **Arctic freshwater lake**  
 925 Sci. Total Environ., 722 (2020), pp. 137904, 10.1016/j.scitotenv.2020.137904  
 926
- 927 Grbić et al., 2020 J. Grbić, P. Helm, S. Athey, C. Rochman  
 928 **Microplastics entering northwestern Lake Ontario are diverse and linked to**  
 929 **urban sources**  
 930 Water Res., 174 (2020), pp. 115623, 10.1016/j.watres.2020.115623  
 931
- 932 Hann et al., 2018 S. Hann, C. Sherrington, O. Jamieson, M. Hickman, P. Kershaw, A.  
 933 Bapasola, G. Cole  
 934 **Investigating options for reducing releases in the aquatic environment of**  
 935 **microplastics emitted by (but not intentionally added in) products**  
 936 Eunomia ICF, Final Report 335  
 937
- 938 Hartmann et al., 2019 N.B. Hartmann, T. Hüffer, R.C. Thompson, M. Hassellöv, A.  
 939 Verschoor, A.E. Daugaard, S. Rist, T. Karlsson, N. Brennholt, M. Cole, M.P.  
 940 Herrling, M.C. Hess, N.P. Ivleva, A.L. Lusher, M. Wagner  
 941 **Are We Speaking the Same Language? Recommendations for a Definition and**  
 942 **Categorization Framework for Plastic Debris**  
 943 Environ. Sci. Technol., 53 (2019), pp. 1039–1047, 10.1021/acs.est.8b05297  
 944
- 945 Hitchcock, 2020 J.N. Hitchcock  
 946 **Storm events as key moments of microplastic contamination in aquatic**  
 947 **ecosystems**  
 948 Sci. Total Envir.  
 949 734 (2020), pp. 139436, 10.1016/j.scitotenv.2020.139436  
 950

- 951 Horton & Dixon, 2018 A.A. Horton, S.J. Dixon  
 952 **Microplastics: an introduction to environmental transport processes**  
 953 WIREs WATER, 5 (2018), pp. e1268, 10.1002/wat2.1268  
 954
- 955 Horton et al., 2017 A.A. Horton, A. Walton, D.J. Spurgeon, E. Lahive, C. Svendsen  
 956 **Microplastics in freshwater and terrestrial environments: Evaluating the current**  
 957 **understanding to identify the knowledge gaps and future research priorities**  
 958 Sci. Total Environ., 586 (2017), pp. 127–141, 10.1016/j.scitotenv.2017.01.190  
 959
- 960 Hu et al., 2020 D. Hu, Y. Zhang, M. Shen  
 961 **Investigation on microplastic pollution of Dongting Lake and its affiliated rivers**  
 962 Mar. Pollut. Bull., 160 (2020), pp. 111555, 10.1016/j.marpolbul.2020.111555  
 963
- 964 Huang et al., 2020 Y. Huang, Q. Liu, W. Jia, C. Yan, J. Wang  
 965 **Agricultural plastic mulching as a source of microplastics in the terrestrial**  
 966 **environment**  
 967 Environ. Pollut., 260 (2020), pp. 114096, 10.1016/j.envpol.2020.114096  
 968
- 969 Hwang et al., 2019 J. Hwang, D. Choi, S. Han, J. Choi, J. Hong  
 970 **An assessment of the toxicity of polypropylene microplastics in human derived**  
 971 **cells**  
 972 Sci. Total Environ., 684 (2019), pp. 657–669, 10.1016/j.scitotenv.2019.05.071  
 973
- 974 Iannilli et al., 2020 V. Iannilli, F. Corami, P. Grasso, F. Lecce, M. Buttinelli, A. Setini  
 975 **Plastic abundance and seasonal variation on the shorelines of three volcanic lakes**  
 976 **in Central Italy: can amphipods help detect contamination?**  
 977 Environ. Sci. Pollut. Res., 27 (2020), pp. 14711–14722, 10.1007/s11356-020-07954-7  
 978
- 979 Imhof et al., 2013 H.K. Imhof, N.P. Ivleva, J. Schmid, R. Niessner, C. Laforsch  
 980 **Contamination of beach sediments of a subalpine lake with microplastic particles**  
 981 Curr. Biol., 23 (2013), pp. PR867–R868, 10.1016/j.cub.2013.09.001  
 982
- 983 INERIS, 2018  
 984 **Données technico-économiques sur les substances chimiques en France :**  
 985 **microplastiques**  
 986 <http://www.ineris.fr/substances/fr/> (last accessed: 01.13.2021)  
 987
- 988 Irfan et al., 2020 T. Irfan, S. Khalid, M. Taneez, M.Z. Hashmi  
 989 **Plastic driven pollution in Pakistan: the first evidence of environmental exposure**  
 990 **to microplastic in sediments and water of Rawal Lake**  
 991 Environ. Sci. Pollut. Res., 27 (2020), pp. 15083-15092, 10.1007/s11356-020-07833-1  
 992
- 993 Jiang et al., 2018 C. Jiang, L. Yin, X. Wen, C. Du, L. Wu, Y. Long, Y. Liu, Y. Ma, Q. Yin, Z.  
 994 Zhou, H. Pan  
 995 **Microplastics in Sediment and Surface Water of West Dongting Lake and South**  
 996 **Dongting Lake: Abundance, Source and Composition**  
 997 Int. J. Environ. Res. Public Health, 15 (2018), pp. 2164, 10.3390/ijerph15102164  
 998
- 999 Julienne et al., 2019 F. Julienne, N. Delorme, F. Lagarde

- 1000 **From macroplastics to microplastics: Role of water in the fragmentation of**  
1001 **polyethylene**  
1002 Chemosphere, 236 (2019), pp. 124409, 10.1016/j.chemosphere.2019.124409  
1003
- 1004 Kaliszewicz et al., 2020 A. Kaliszewicz, M. Winczek, K. Karaban, D. Kurzydłowski, M.  
1005 Górską, W. Koselak, J. Romanowski  
1006 **The contamination of inland waters by microplastic fibres under different**  
1007 **anthropogenic pressure: Preliminary study in Central Europe (Poland)**  
1008 Waste Manag. Res., 38 (2020), pp. 1231-1238, 10.1177/0734242X20938448  
1009
- 1010 Karnaukhov et al., 2020 D. Karnaukhov, S. Biritskaya, E. Dolinskaya, M. Teplykh, N.  
1011 Silenko, Y. Ermolaeva, E. Silow  
1012 **Pollution by macro- and microplastic of large lacustrine ecosystems in Eastern**  
1013 **Asia**  
1014 Pollut. Res., 39 (2020), pp. 353-355  
1015
- 1016 Kirstein et al., 2016 I.V. Kirstein, S. Kirmizi, A. Wichels, A. Garin-Fernandez, R. Erler, M.  
1017 Löder, G. Gerdts  
1018 **Dangerous hitchhikers? Evidence for potentially pathogenic Vibrio spp. on**  
1019 **microplastic particles**  
1020 Mar. Environ. Res., 120 (2016), pp. 1–8, 10.1016/j.marenvres.2016.07.004  
1021
- 1022 Klein & Fischer, 2019 M. Klein, E.K. Fischer  
1023 **Microplastic abundance in atmospheric deposition within the Metropolitan area**  
1024 **of Hamburg, Germany**  
1025 Sci. Total Environ., 685 (2019), pp. 96–103, 10.1016/j.scitotenv.2019.05.405  
1026
- 1027 Koelmans et al., 2019 A.A. Koelmans, N.H. Mohamed Nor, E. Hermsen, M. Kooi, S.M.  
1028 Mintenig, J. De France  
1029 **Microplastics in freshwaters and drinking water: Critical review and assessment**  
1030 **of data quality**  
1031 Water Res., 155 (2019), pp. 410–422, 10.1016/j.watres.2019.02.054  
1032
- 1033 Lares et al., 2018 M. Lares, M. Chaker Ncibi, Ma. Sillanpää, Mi. Sillanpää  
1034 **Occurrence, identification and removal of microplastic particles and fibers in**  
1035 **conventional activated sludge process and advanced MBR technology**  
1036 Water Res., 133 (2018), pp. 236–246, 10.1016/j.watres.2018.01.049  
1037
- 1038 Lebreton & Andrady 2019 L. Lebreton, A. Andrady  
1039 **Future scenarios of global plastic waste generation and disposal**  
1040 Palgrave Communications, 5, 6, 10.1057/s41599-018-0212-7  
1041
- 1042 Li et al., 2020 C. Li, R. Busquets, L.C. Campos  
1043 **Assessment of microplastics in freshwater systems: A review**  
1044 Sci. Total Environ., 707 (2020), pp. 135578, 10.1016/j.scitotenv.2019.135578  
1045
- 1046 Mason et al., 2020 S.A. Mason, J. Daily, G. Aleid, R. Ricotta, M. Smith, K. Donnelly, R.  
1047 Knauff, W. Edwards, M.J. Hoffman  
1048 **High levels of pelagic plastic pollution within the surface waters of Lakes Erie**  
1049 **and Ontario**

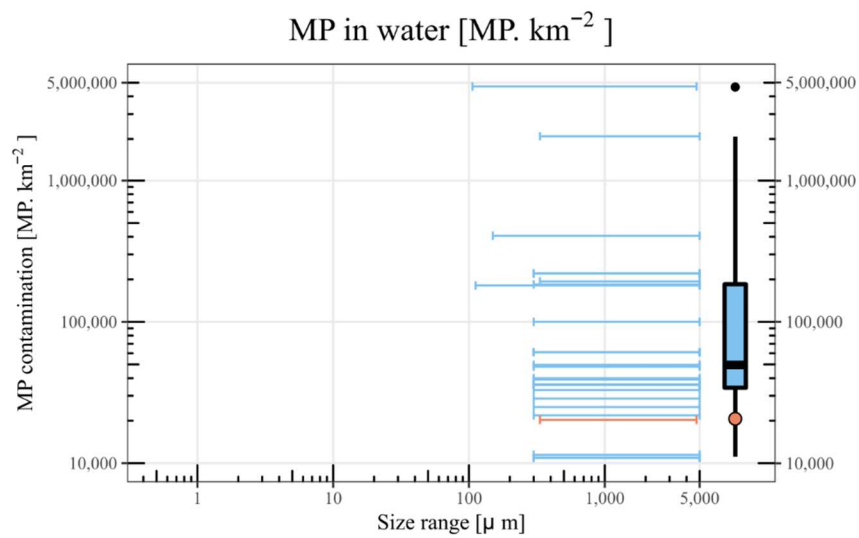
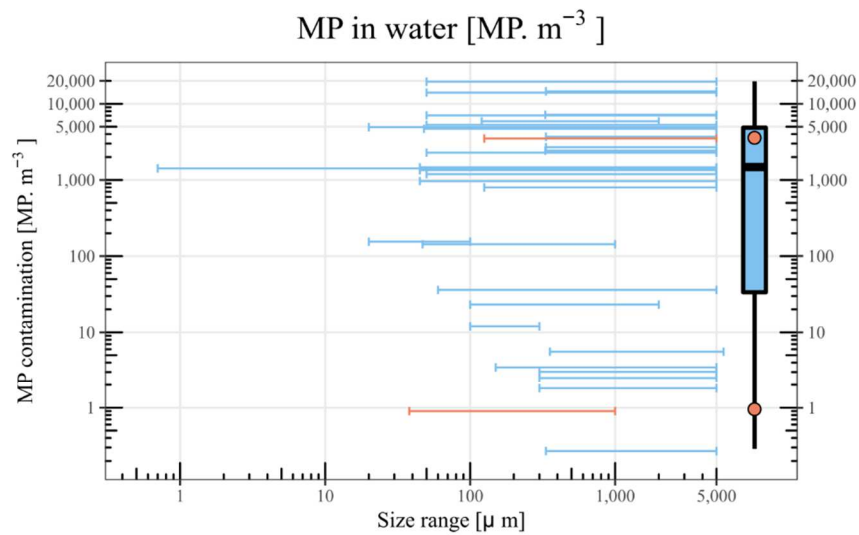
1050 J. Great Lakes Res., 46 (2020), pp. 277-288, 10.1016/j.jglr.2019.12.012  
1051  
1052 Mason et al., 2016 S.A. Mason, L. Kammin, M. Eriksen, G. Aleid, S. Wilson, C. Box, N.  
1053 Williamson, A. Riley  
1054 **Pelagic plastic pollution within the surface waters of Lake Michigan, USA**  
1055 J. Great Lakes Res., 42 (2016), pp. 753–759, 10.1016/j.jglr.2016.05.009  
1056  
1057 Meng et al., 2020 Y. Meng, F.J. Kelly, S.L. Wright  
1058 **Advances and challenges of microplastic pollution in freshwater ecosystems: A**  
1059 **UK perspective**  
1060 Environ. Pollut., 256 (2020), pp. 113445, 10.1016/j.envpol.2019.113445  
1061  
1062 Merga et al., 2020 L.B. Merga, P.E. Redondo-Hasselerharm, P.J. Van den Brink, A.A.  
1063 Koelmans  
1064 **Distribution of microplastic and small macroplastic particles across four fish**  
1065 **species and sediment in an African lake**  
1066 Sci. Total Environ., 741 (2020), pp. 140527, 10.1016/j.scitotenv.2020.140527  
1067  
1068 Migwi et al., 2020 F.K. Migwi, J.A. Ogunah, J.M. Kiratu  
1069 **Occurrence and Spatial Distribution of Microplastics in the surface Waters of**  
1070 **Lake Naivasha, Kenya**  
1071 Environ. Chem., 39 (2020), pp. 765-774, 10.1002/etc.4677  
1072  
1073 Mintenig et al., 2017 S.M. Mintenig, I. Int-Veen, M.G.J. Löder, S. Primpke, G. Gerds, G.  
1074 **Identification of microplastic in effluents of waste water treatment plants using**  
1075 **focal plane array-based micro-Fourier-transform infrared imaging**  
1076 Water Res., 108 (2017), pp. 365-372, 10.1016/j.watres.2016.11.015  
1077  
1078 Montuelle & Clémens, 2015 B. Montuelle, A. Clémens  
1079 **Le tour des grands lacs alpins en 80 questions**  
1080 Graie publishing, pp. 4-205  
1081  
1082 Negrete Velasco et al., 2020 A. de J. Negrete Velasco, L. Rard, W. Blois, D. Lebrun, F.  
1083 Lebrun, F. Pothe, S. Stoll  
1084 **Microplastic and Fibre Contamination in a Remote Mountain Lake in**  
1085 **Switzerland**  
1086 Water, 12 (2020), pp. 2410, 10.3390/w12092410  
1087  
1088 Oni et al., 2020 B.A. Oni, A.O. Ayeni, O. Agboola, T. Oguntade, O. Obanla  
1089 **Comparing microplastics contaminants in (dry and raining) seasons for Ox- Bow**  
1090 **Lake in Yenagoa, Nigeria**  
1091 Ecotoxicol. Environ. Saf., 198 (2020), pp. 110656, 10.1016/j.ecoenv.2020.110656  
1092  
1093 Pastorino et al., 2020 P. Pastorino, E. Pizzul, M. Bertoli, S. Anselmi, M. Kušće, V. Menconi,  
1094 M. Prearo, M. Renzi  
1095 **First insights into plastic and microplastic occurrence in biotic and abiotic**  
1096 **compartments, and snow from a high-mountain lake (Carnic Alps)**  
1097 Chemosphere, 265 (2020), pp. 129121, 10.1016/j.chemosphere.2020.129121  
1098

- 1099 Picó et al., 2020 Y. Picó, R. Alvarez-Ruiz, A.H. Alfarhan, M.A. El-Sheikh, H.O. Alshahrani,  
1100 D. Barceló  
1101 **Pharmaceuticals, pesticides, personal care products and microplastics**  
1102 **contamination assessment of Al-Hassa irrigation network (Saudi Arabia) and its**  
1103 **shallow lakes**  
1104 Sci. Total Environ., 701 (2020), pp.135021, 10.1016/j.scitotenv.2019.135021  
1105
- 1106 PlasticsEurope, 2019  
1107 **Plastics - the Facts 2019: an analysis of European plastics production, demand**  
1108 **and waste data**  
1109 <https://www.plasticseurope.org/fr/resources/market-data> (last accessed: 01.13.2021)  
1110
- 1111 PlasticsEurope, 2011  
1112 **Plastics - the Facts 2011: an analysis of European plastics production, demand**  
1113 **and recovery for 2010**  
1114 <https://www.plasticseurope.org/fr/resources/market-data> (last accessed: 01.13.2021)  
1115
- 1116 Sarker et al., 2020 A. Sarker, D.M. Deepo, R. Nandi, J. Rana, S. Islam, S. Rahman, M.N.  
1117 Hossain, Md.S. Islam, A. Baroi, J.-E. Kim  
1118 **A review of microplastics pollution in the soil and terrestrial ecosystems: A global**  
1119 **and Bangladesh perspective**  
1120 Sci. Total Environ., 733 (2020), pp. 139296, 10.1016/j.scitotenv.2020.139296  
1121
- 1122 Scopetani et al., 2019 C. Scopetani, D. Chelazzi, A. Cincinelli, M. Esterhuizen-Londt  
1123 **Assessment of microplastic pollution: occurrence and characterisation in**  
1124 **Vesijärvi lake and Pikku Vesijärvi pond, Finland**  
1125 Environ. Monitor. Assess., 191 (2019), pp. 652, 10.1007/s10661-019-7843-z  
1126
- 1127 Sighicelli et al., 2018 M. Sighicelli, L. Pietrelli, F. Lecce, V. Iannilli, M. Falconieri, L.  
1128 Coscia, S. Di Vito, S. Nuglio, G. Zampetti  
1129 **Microplastic pollution in the surface waters of Italian Subalpine Lakes**  
1130 Environ. Pollut., 236 (2018), pp. 645–651, 10.1016/j.envpol.2018.02.008  
1131
- 1132 Song et al., 2017 Y.K. Song, S.H. Hong, M. Jang, G.M. Han, S.W. Jung, W.J. Shim  
1133 **Combined effects of UV exposure duration and mechanical abrasion on**  
1134 **microplastic fragmentation by polymer type**  
1135 Environ. Sci. Technol., 51 (2017), pp. 4368–4376, 10.1021/acs.est.6b06155  
1136
- 1137 Stark, 2019 M. Stark  
1138 **Letter to the Editor Regarding “Are We Speaking the Same Language?**  
1139 **Recommendations for a Definition and Categorization Framework for Plastic**  
1140 **Debris”**  
1141 Environ. Sci. Technol., 53 (2019), pp. 4677–4677, 10.1021/acs.est.9b01360  
1142
- 1143 Su et al., 2016 L. Su, Y. Xue, L. Li, D. Yang, P. Kolandhasamy, D. Li, H. Shi  
1144 **Microplastics in Taihu Lake, China**  
1145 Environ. Pollut., 216 (2016), pp. 711–719, 10.1016/j.envpol.2016.06.036  
1146
- 1147 Talvitie et al., 2017 J. Talvitie, A. Mikola, O. Setälä, M. Heinonen, A. Koistinen

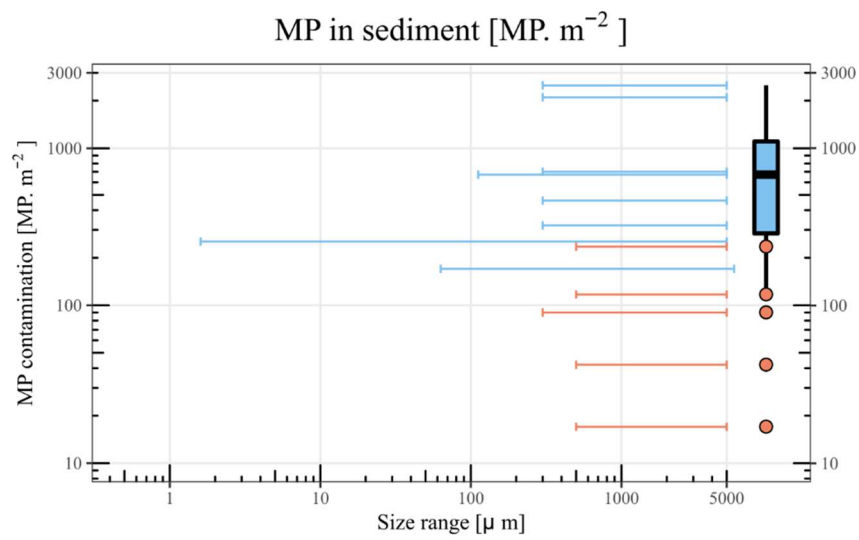
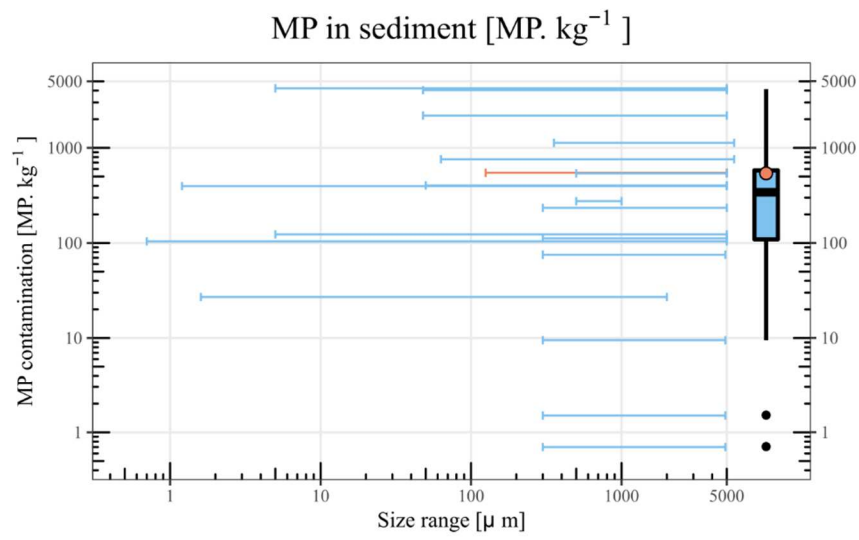
- 1148           **How well is microlitter purified from wastewater? – A detailed study on the**  
1149           **stepwise removal of microlitter in a tertiary level wastewater treatment plant**  
1150           Water Res., 109 (2017), pp. 164–172, 10.1016/j.watres.2016.11.046  
1151
- 1152           Turner et al., 2019 S. Turner, A.A. Horton, N.L. Rose, C. Hall  
1153           **A temporal sediment record of microplastics in an urban lake, London, UK**  
1154           J. Paleolimnol., 61 (2019), pp. 449–462, 10.1007/s10933-019-00071-7  
1155
- 1156           Uddin et al., 2020 S. Uddin, S. Fowler, M. Behbehani  
1157           **An Assessment of Microplastic Inputs into the Aquatic Environment from**  
1158           **Wastewater Streams**  
1159           Mar. Pollut. Bull., 160 (2020), pp. 111538, 10.1016/j.marpolbul.2020.111538  
1160
- 1161           United Nations, 2020  
1162           **Sustainable Development Goals Knowledge Platform**  
1163           <https://sustainabledevelopment.un.org/> (last accessed 4.29.20)  
1164
- 1165           Uurasjärvi et al., 2020 E. Uurasjärvi, S. Hartikainen, O. Setälä, M. Lehtiniemi, A. Koistinen  
1166           **Microplastic concentrations, size distribution, and polymer types in the surface**  
1167           **waters of a northern European lake**  
1168           Water Envir. Res., 92 (2020), pp. 149–156, 10.1002/wer.1229  
1169
- 1170           Vaughan et al., 2017 R. Vaughan, S.D. Turner, N.L. Rose  
1171           **Microplastics in the sediments of a UK urban lake**  
1172           Environ. Pollut., 229 (2017), pp. 10–18, 10.1016/j.envpol.2017.05.057  
1173
- 1174           Wagner et al., 2018 S. Wagner, T. Hüffer, P. Klöckner, M. Wehrhahn, T. Hofmann, T.  
1175           Reemtsma  
1176           **Tire wear particles in the aquatic environment - A review on generation, analysis,**  
1177           **occurrence, fate and effects**  
1178           Water Res., 139 (2018), pp. 83–100, 10.1016/j.watres.2018.03.051  
1179
- 1180           Wang et al., 2017 W. Wang, A.W. Ndungu, Z. Li, J. Wang  
1181           **Microplastics pollution in inland freshwaters of China: A case study in urban**  
1182           **surface waters of Wuhan, China**  
1183           Sci. Total Envir., 575 (2017), pp. 1369–1374, 10.1016/j.scitotenv.2016.09.213  
1184
- 1185           Wang et al., 2019 Z. Wang, Y. Qin, W. Li, W. Yang, Q. Meng, J. Yang  
1186           **Microplastic contamination in freshwater: first observation in Lake Ulansuhai,**  
1187           **Yellow River Basin, China**  
1188           Environ. Chem. Lett., 17 (2019), pp. 1821–1830, 10.1007/s10311-019-00888-8  
1189
- 1190           Welden & Lusher, 2017 N.A. Welden, A.L. Lusher  
1191           **Impacts of changing ocean circulation on the distribution of marine microplastic**  
1192           **litter: changing ocean circulation and marine microplastic litter**  
1193           Integr. Environ. Assess. Manag., 13 (2017), pp. 483–487, 10.1002/ieam.1911  
1194
- 1195           Wong et al., 2020 J.K.H. Wong, K.K. Lee, K.H.D. Tang, P.-S. Yap  
1196           **Microplastics in the freshwater and terrestrial environments: prevalence, fates,**  
1197           **impacts and sustainable solutions**

- 1198 Sci. Total Environ., 719 (2020), pp. 137512, 10.1016/j.scitotenv.2020.137512  
 1199
- 1200 Wright et al., 2020 S.L. Wright, J. Ulke, A. Font, K.L.A. Chan, F.J. Kelly  
 1201 **Atmospheric microplastic deposition in an urban environment and an evaluation**  
 1202 **of transport**  
 1203 Environ. Int., 136 (2020), pp. 105411, 10.1016/j.envint.2019.105411  
 1204
- 1205 Wu et al., 2019 X. Wu, J. Pan, M. Li, Y. Li, M. Bartlam, Y. Wang  
 1206 **Selective enrichment of bacterial pathogens by microplastic biofilm**  
 1207 Water Res., 165 (2019), pp. 114979, 10.1016/j.watres.2019.114979  
 1208
- 1209 Xia et al., 2020 W. Xia, Q. Rao, X. Deng, J. Chen, P. Xie  
 1210 **Rainfall is a significant environmental factor of microplastic pollution in inland**  
 1211 **waters**  
 1212 Sci. Total Environ., 732 (2020), pp. 139065, 10.1016/j.scitotenv.2020.139065  
 1213
- 1214 Xiong et al., 2018 X. Xiong, K. Zhang, X. Chen, H. Shi, Z. Luo, C. Wu  
 1215 **Sources and distribution of microplastics in China's largest inland lake – Qinghai**  
 1216 **Lake**  
 1217 Environ. Pollut., 235 (2018), pp. 899-906, 10.1016/j.envpol.2017.12.081  
 1218
- 1219 Yin et al., 2020 L. Yin, X. Wen, C. Du, J. Jiang, L. Wu, Y. Zhang, Z. Hu, S. Hu, Z. Feng, Z.  
 1220 Zhou, Y. Long, Q. Gu  
 1221 **Comparison of the abundance of microplastics between rural and urban areas: A**  
 1222 **case study from East Dongting Lake**  
 1223 Chemosphere, 244 (2020), pp. 125486, 10.1016/j.chemosphere.2019.125486  
 1224
- 1225 Yuan et al., 2019 W. Yuan, X. Liu, W. Wang, M. Di, J. Wang  
 1226 **Microplastic abundance, distribution and composition in water, sediments, and**  
 1227 **wild fish from Poyang Lake, China**  
 1228 Ecotoxicol. Environ. Safety, 170 (2019), pp. 180–187, 10.1016/j.ecoenv.2018.11.126  
 1229
- 1230 Zbyszewski & Corcoran, 2011 M. Zbyszewski, P.L. Corcoran  
 1231 **Distribution and Degradation of Fresh Water Plastic Particles Along the Beaches**  
 1232 **of Lake Huron, Canada**  
 1233 Water Air Soil Pollut., 220 (2011), pp. 365–372, 10.1007/s11270-011-0760-6  
 1234
- 1235 Zbyszewski et al., 2014 M. Zbyszewski, P.L. Corcoran, A. Hockin  
 1236 **Comparison of the distribution and degradation of plastic debris along shorelines**  
 1237 **of the Great Lakes, North America**  
 1238 J. Great Lakes Res., 40 (2014), pp. 288–299, 10.1016/j.jglr.2014.02.012  
 1239
- 1240 Zhang et al., 2016 K. Zhang, J. Su, X. Xiong, X. Wu, C. Wu, J. Liu  
 1241 **Microplastic pollution of lakeshore sediments from remote lakes in Tibet plateau,**  
 1242 **China**  
 1243 Environ. Pollut., 219 (2016), pp. 450–455, 10.1016/j.envpol.2016.05.048  
 1244
- 1245 Zhang et al., 2020 Q. Zhang, E.G. Xu, J. Li, Q. Chen, L. Ma, E.Y. Zeng, H. Shi  
 1246 **A Review of Microplastics in Table Salt, Drinking Water, and Air: Direct Human**  
 1247 **Exposure**

- 1248 Environ. Sci. Technol., 54 (2020), pp. 3740-3751, 10.1021/acs.est.9b04535  
1249
- 1250 Zhang et al., 2019 Y. Zhang, T. Gao, S. Kang, M. Sillanpää  
1251 **Importance of atmospheric transport for microplastics deposited in remote areas**  
1252 Environ. Pollut., 254 (2019), pp. 112953, 10.1016/j.envpol.2019.07.121  
1253
- 1254 S. Zhang et al., 2020 S. Zhang, S. Yulan, S. Kang, S. Allen, D. Allen, T. Gao, M. Sillanpää  
1255 **Atmospheric microplastics: A review on current status and perspectives**  
1256 Earth Sci. Rev., 203 (2020), pp. 103118, 10.1016/j.earscirev.2020.103118  
1257
- 1258 Y. Zhang et al., 2020, Y. Zhang, J. Liang, G. Zeng, W. Tang, Y. Lu, Y. Luo, W. Xing, N.  
1259 Tang, S. Ye, X. Li, W. Huang  
1260 **How climate change and eutrophication interact with microplastic pollution and**  
1261 **sediment resuspension in shallow lakes: a review**  
1262 Sci. Total Environ., 705 (2020), pp. 135979, 10.1016/j.scitotenv.2019.135979  
1263
- 1264 Zhou et al., 2020 Q. Zhou, C. Tian, Y. Luo  
1265 **Various forms and deposition fluxes of microplastics identified in the coastal**  
1266 **urban atmosphere**  
1267 Chin. Sci. Bull., 62 (2020), pp. 3902–3909, 10.1360/N972017-00956  
1268

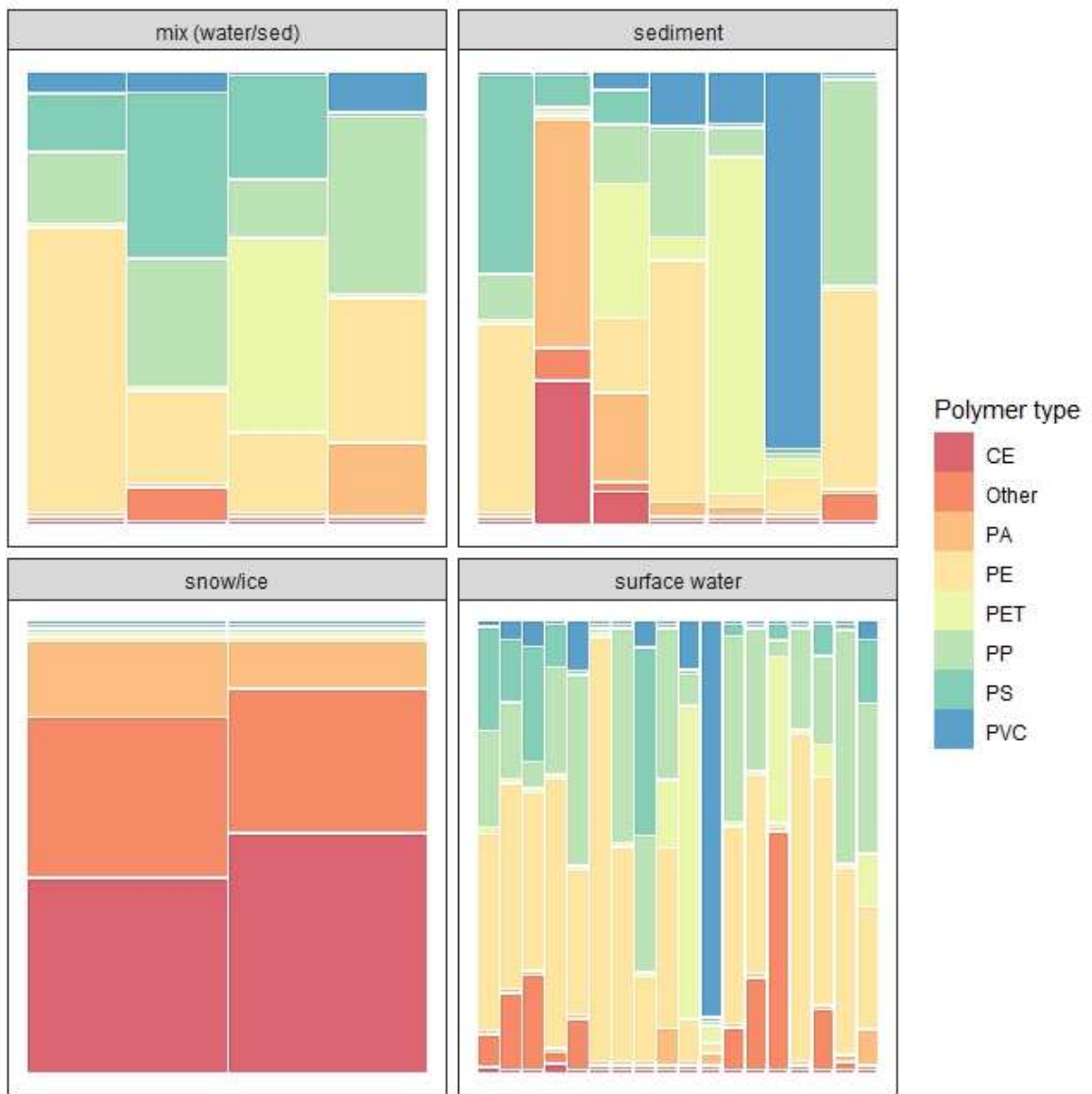


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 2 Fig. 1. Microplastic concentrations on surface waters of worldwide lakes ( $\text{MP}/\text{m}^3$  or  $\text{MP}/\text{km}^2$ ) according to size  
 3 range studied and anthropogenic pressure (rural, urban)



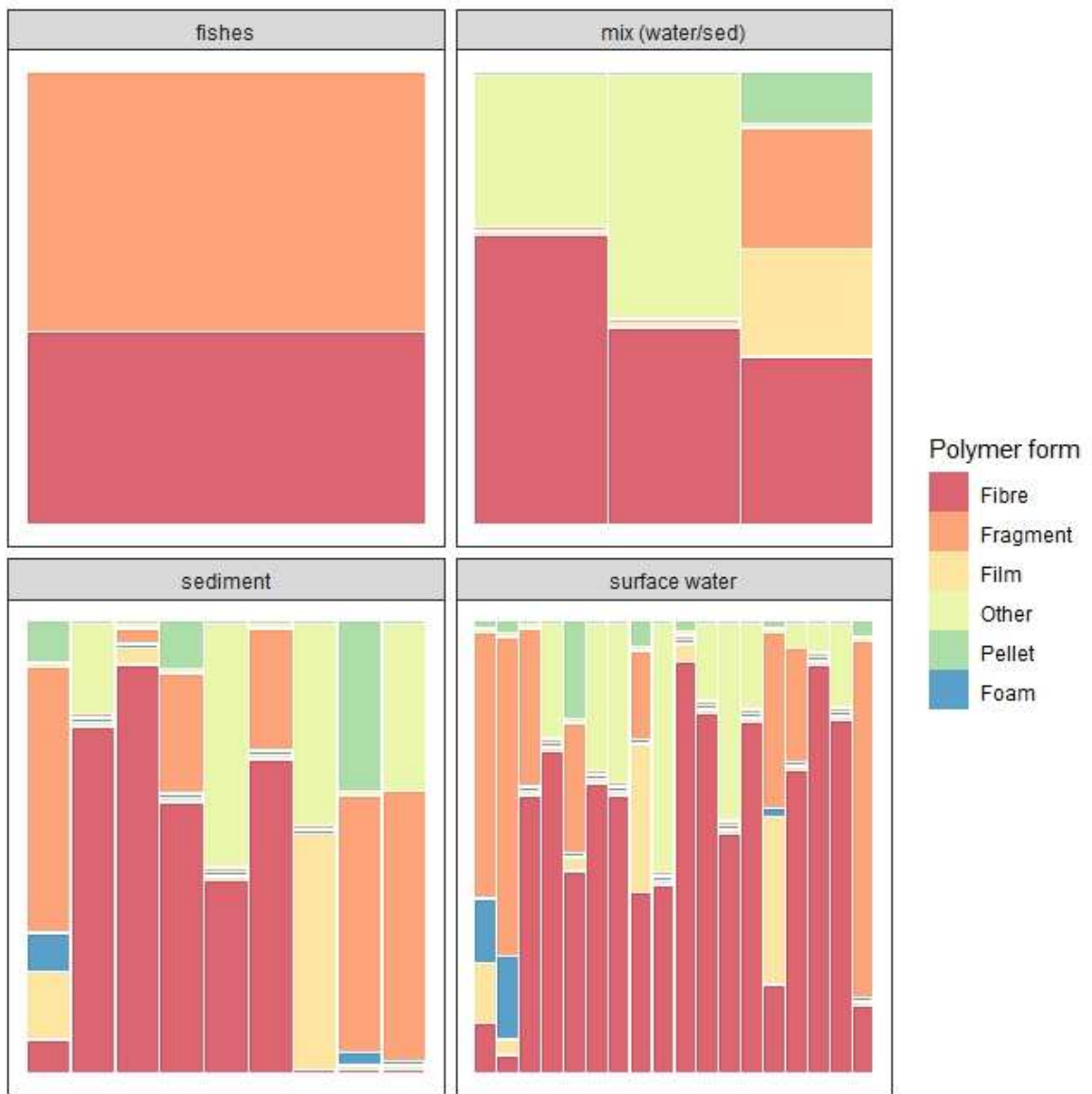
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5 Fig. 2. Microplastic concentrations in overall sediments (LBoS, LBeS, LShS) of worldwide lakes (MP/kg or  
 6 MP/m<sup>2</sup>) according to size range studied and anthropogenic pressure (rural, urban)



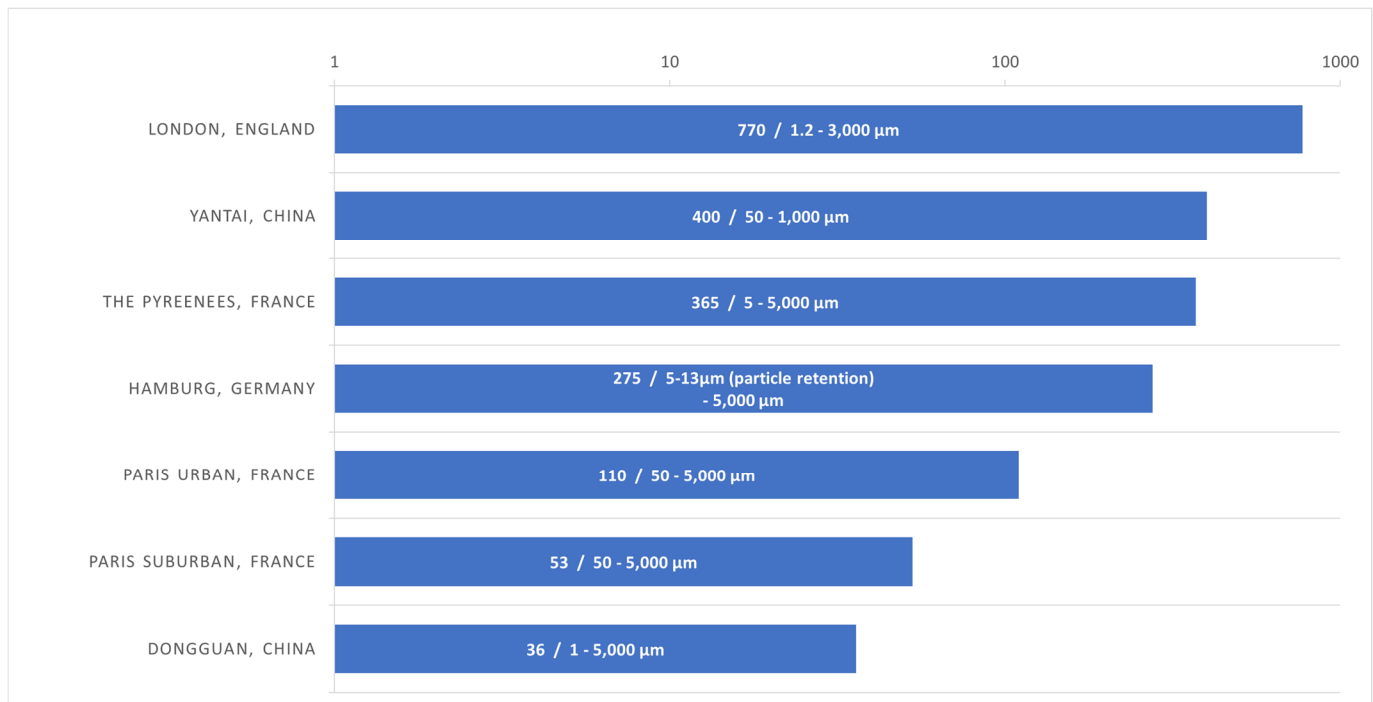
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8 Fig. 3. Distribution of the polymer types in all the compartments of the lacustrine environments across the globe



9

10 Fig. 4. Distribution of the particle forms in all the compartments of the lacustrine environments across the globe



11

12 Fig. 5. Atmospheric fallout (wet and dry) of microplastics between different geographical localization  
 13 (particle/km<sup>2</sup>/day x 10<sup>6</sup>). For each localization, mean concentration is written and also the size class studied  
 14 (μm). Numbers are rounded to the nearest unit.

Country	Mesh ( $\mu\text{m}$ )	Size ( $\mu\text{m}$ )	Concentration on surface water	References
Argentina	38 (N)	70.7 % < 1,000	$9.0 \times 10^{-1} \pm 6.0 \times 10^{-1} \text{ mp/m}^3$	Alfonso et al., 2020
	47 (N)	62.8 % < 1,000	$1.4 \times 10^2 \pm 4.0 \times 10^1 \text{ mp/m}^3$	Alfonso et al., 2020
China	48 (P)	79.8 % < 1,000	$4.7 \times 10^3 \pm 2.8 \times 10^3 \text{ mp/m}^3$	Di et Wang, 2018
	50 (P)	80 % < 2,000	$1.7 \times 10^3 \pm 6.4 \times 10^2 \text{ mp/m}^3$	Wang et al., 2017
	50 (P)	65 % < 2,000	$1.2 \times 10^2 \pm 2.3 \times 10^3 \text{ mp/m}^3$	Wang et al., 2018
	112 (N)	73.1 % < 500	$5.0 - 3.4 \times 10^1 \text{ mp/m}^3$	Yuan et al., 2019
Italy	300 (N)	< 500	2.5 and $3.0 \text{ mp/m}^3$ (two lakes)	Fischer et al., 2016
China	330 (N)	98.2 % < 2,000	$3.1 - 1.1 \times 10^1 \text{ mp/m}^3$	Mao et al., 2020
Finland	333 (N) and 20; 100; 300 (P)	20 - 300	$0.27 \pm 0.18$ and $1.6 \times 10^2 \pm 73$ ; $12 \pm 17$ ; $1.8 \pm 2.3 \text{ mp/m}^3$ respectively	Uurasjärvi et al., 2020
Canada, USA	355 (N)	81 % < 999	$4.3 \times 10^4 \pm 1.2 \times 10^5 \text{ mp/km}^2$	Eriksen et al., 2013
	355 (N)	59 % < 999	$1.7 \times 10^4 \text{ mp/km}^2$	Mason et al., 2016
	355 (N)	73 % < 999	$2.3 \times 10^5$ and $4.5 \times 10^4 \text{ mp/km}^2$ (two lakes)	Mason et al., 2020
Mongolia	355 (N)	41 % < 999	$1.3 \times 10^4 \text{ mp/km}^2$	Free et al., 2014

1

2 Table 1: five-columns

3 Table 1. Distribution of the plastic particle sizes in worldwide lakes with their concentration on the surface water according to sampling  
4 methodology used. (N) means researches used net and (P) means researches used pump.

5

Colors should be used for the graphical abstract

Graphical abstract

