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1 Proposed title: 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/km², 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 μm 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 ± 0.18 particles/ m^3 obtained with a 333 μm Manta net and 1.80
121 ± 2.30 particles/ m^3 obtained with a 300 μm 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 μm net is the most commonly used (particularly 300, 333, and 355
130 μm). Approximately 80 % of 50 published aquatic surveys concerning MPs in aquatic
131 environments only accounted for plastics ≥ 300 μm in diameter, whereas > 95 % of particles

132 in surveyed products were smaller than the 300 μm minimum diameter (Conkle et al., 2018).
133 The use of 50 μm 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 μm (Kaliszewicz et al., 2020),
136 20 μm (Bharath K et al., 2021), 38 μm (Alfonso et al., 2020b), 50 μm (Pastorino et al., 2020)
137 and, 60 μm (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 μm and 300 μm 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 ~ 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/m³ 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/m³), 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 μm , 50 μm -300 μm , 300 μm -1 mm,
205 1 mm-5 mm. Moreover, the detection limit of FTIR is approximately 6.25 μm , we suggest to

206 set the limit at 8 times the measurement limit. We suggest a limit of 50 μm because studies
207 that worked on small plastic particles were at or below this size (Section 5.1), including 38
208 μm (Alfonso et al., 2020b), 45 μm (Jiang et al., 2018), 47 μm (Alfonso et al., 2020a), and 50
209 μm (Wang et al., 2017; Xia et al., 2020; Yuan et al., 2019), for surface water. Several studies
210 worked at or above the 100 μm limit for surface water, namely 100 μm (Bordós et al., 2019),
211 106 μm (Cable et al., 2017), 112 μm (Xiong et al., 2018), 125 μm (Grbić et al., 2020), and
212 150 μm (Migwi et al., 2020). The 300 μm frontier corresponds to the size of a plankton net,
213 and so far, most studies have worked with 300 μm or 333 μm nets. Among studies concerning
214 MPs in lakes, several quantified particles up to 999 μm (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 (m^3/s), the water retention time (year), the
226 surface area (km^2), volume (km^3), mean and maximum depth (m), watershed area (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/ 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/m³, MP/km², MP/kg and, MP/m²) and
252 available data. Concerning surface waters (**Fig. 1**), values were comprised between 0.27
253 MP/m³ and 34,000 MP/m³ with a median value around 1,442 MP/m³ and, between 10,900

254 MP/km² and about 4,700,000 MP/km² with a median value of 48,200 MP/km². 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/m³, 4,000 MP/m³ and, 20,000 MP/km² confirming the
257 importance of investigating more rural and remote lakes. MPs in water showed larger size
258 ranges for MP/m³ than MP/km², 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/m²
265 and 2,500 MP/m² with a median value around 290 MP/m². 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/km² with a size class of 333 μ 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/m³ (Karnaikhov et al., 2020). Nine lakes in the Pampean Region (Argentina) contained

278 0.3 MP/m³ to 1.9 MP/m³ with a mean of 0.9 ± 0.6 MP/m³ and a size class between of 38 μ m
279 to 1 mm (Alfonso et al., 2020b). Finally, Lake Sassolo (Switzerland) contained mean
280 concentrations of 2.6×10^3 MP/m³ and 4.4×10^3 fibres/m³ (natural and synthetic fibres were
281 excluded from MPs) (Negrete Velasco et al., 2020), 4.1 μ g of MP fibres/m² 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/m³) is almost 8 times deeper and 55,200 times larger (on the surface) than
286 Sassolo Lake (2.6×10^3 MP/m³). 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/km² (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/km² for Siling Co, $42 \pm 47 \times 10^6$ particles/km² for Geren Co, $117 \pm 126 \times 10^6$
297 particles/km² for Wuru Co, and $17 \pm 20 \times 10^6$ particles/km² 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/m³ (from 5,333 particles/m³ to 97,333 particles/m³ 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/km² to 2,090,000 MP/km² or from 0.27 MP/m³ to 34,000 MP/m³ (**Table S2**). For
314 instance, concentrations of $180,900 \pm 229,533$ MP/km² with a size class ranging from 112 µm
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 3.1×10^3 MP/m³ and 11.3×10^3 MP/m³ with a
317 size class from 330 µm to 5 mm was reported for Ulansuhai Nur Lake (China) (Wang et al.,
318 2019). In Lake Dziekanowskie and Kalwa Lake (Poland), 4.7 µg of MP fibres/m³ and 4.4 µg
319 of MP fibres/m³ 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/km² (Faure et al., 2015), Iseo (Italy) with 11,479.00 MP/km² (Binelli et al.,
322 2020), Michigan (USA) with 15,698.00 MP/km² (Mason et al., 2016), and Kallavesi (Finland)
323 with 0.27 ± 0.18 MP/m³ (Uurasjärvi et al., 2020), Bolsena (Italy) with 2.51 MP/m³, Chiusi
324 (Italy) with 3.02 MP/m³ (Fischer et al., 2016) and, Simcoe (Canada) with 0.4 – 1.3 MP/m³
325 (Felismino et al., 2020 In Press, Corrected Proof). Therefore, some rural lakes (such as
326 Khövsgöl with 20,264 MP/km² or Sassolo with 2.6×10^3 MP/m³) seem more contaminated

327 than some urban lakes. The urban Kallavesi Lake (Finland) contains 0.27 ± 0.18 MP/m³
328 (Uurasjärvi et al., 2020), whereas nine rural lakes in Argentina contain 0.90 ± 0.60 MP/m³
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 km²), compared with that of Argentinean lakes (mean of 88 km²) could be a plausible
332 explanation for the MP concentration. The different sampling methodologies, with a mesh
333 size of 333 μ m in Finland and 38 μ 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/km² and 20,264 particles/km², respectively. However,
337 these disparities could also be linked to the sampling methodology because 112 μ m and 333
338 μ 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/km² or particles/m³ 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/m² to 339.0 MP/m². The
343 concentrations of lakeshore sediments vary less, from 104 MP/kg to 760 MP/kg. Dongting
344 Lake (Yueyang, China) contained 389 ± 66 particles/m³ to 501 ± 331 particles/m³ in the
345 lakeshore sediments (Jiang et al., 2018). The MP concentrations in sediments of Lake
346 Yenagoa (Nigeria) ranged from 1,000 particles/m³ to 8,330 particles/m³ during the dry season
347 and 201 particles/m³ to 8,370 particles/m³ during the raining season (Oni et al., 2020). Finally,
348 the lakeshore sediments of lakes Albano, Bracciano, and Vico (Italy) contained 3.4 g/m² 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/km² in the

352 surface water and 50 particles/km² to 1,290 × 10⁶ particles/km² 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/m³, particles/m², particles/g, and g/m²). 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 × 10³ particles/m³ melted snow and 8 ± 1 × 10³
375 particles/m³ 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 μm
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 μm –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 μm to 300 μm 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 μm , 50 % < 670 μm , and 75 % < 1,400 μm
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 μm) than
400 those in surface water and sediments (73.1 % and 57.1% < 500 μm , 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 μm or pumped through different filters: 20, 48, 50, 100, 300 μm . Further studies must be
411 conducted using a unique sampling methodology, unit system (particles/ km^2 or particles/ 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 ± 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/km² (Grand
546 Lac) and 27,000 mg/km² (Petit Lac) (Faure et al., 2015) we estimated a fragmentation flux
547 from the surface as high as 1.1×10^8 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/km²) in surface waters of lakes,
550 namely 2,808.98 macroplastics/km² (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/km². 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 ± 6.0 particles/L to 12.8 ± 5.8
564 particles/L with intra-day variations of 6.6×10^8 particles/d and 1.2×10^9 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 ± 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 3.9×10^{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/m³ to 17,383 particles/m³ 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 ± 7.9 particles/L in stormwater runoff
588 (sampling in urban rivers during wet weather) and 0.9 ± 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 ± 49.3 , 308.0 ± 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, 3.5×10^{10} and 7.6×10^{10} synthetic fibres could
618 originate from the atmosphere per year, thereby leading to an atmospheric flux of 18×10^6
619 synthetic fibres/(km²·d) to 37×10^6 synthetic fibres/(km²·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 36×10^6
627 particles/(km²·d) to 770×10^6 particles/(km²·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 287×10^6
630 particles/(km²·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/m³ in stormwater runoff (relative abundances of fibres

644 of 41 %, and TRWPs of 22 %), $13,300 \pm 15,500$ particles/m³ in wastewater effluent (90 %
645 fibres), and $900 \pm 1,300$ particles/m³ 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 1.1×10^8
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 μm 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. Dagaard, 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 Górska, 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**

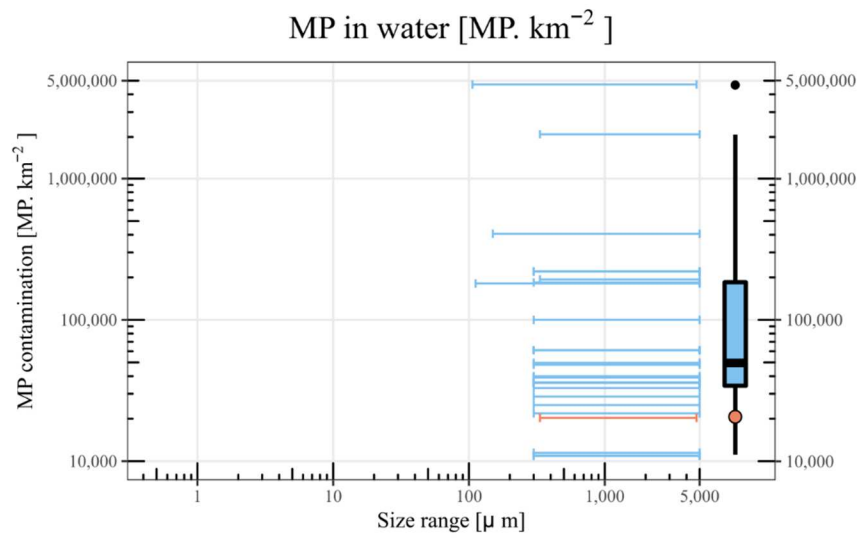
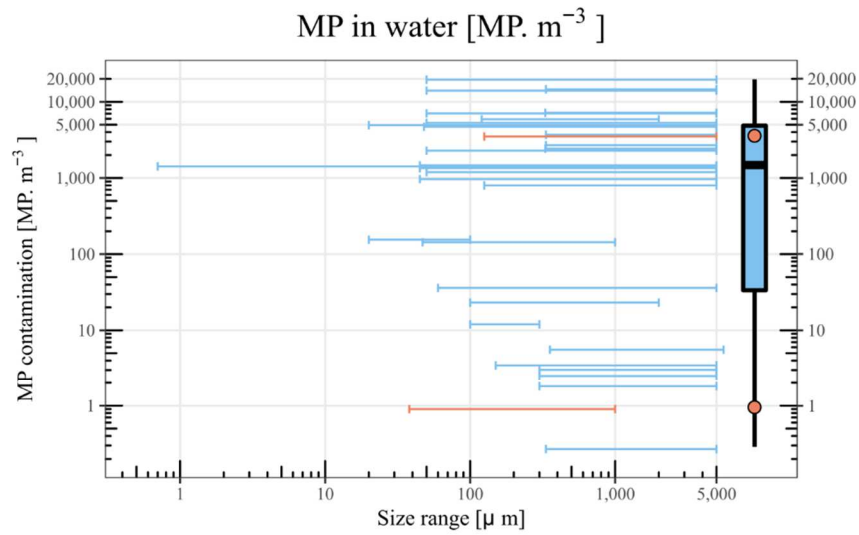
1050 J. Great Lakes Res., 46 (2020), pp. 277-288, 10.1016/j.jglr.2019.12.012
1051
1052 Mason et al., 2016 S.A. Mason, L. Kammin, M. Eriksen, G. Aleid, S. Wilson, C. Box, N.
1053 Williamson, A. Riley
1054 **Pelagic plastic pollution within the surface waters of Lake Michigan, USA**
1055 J. Great Lakes 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**

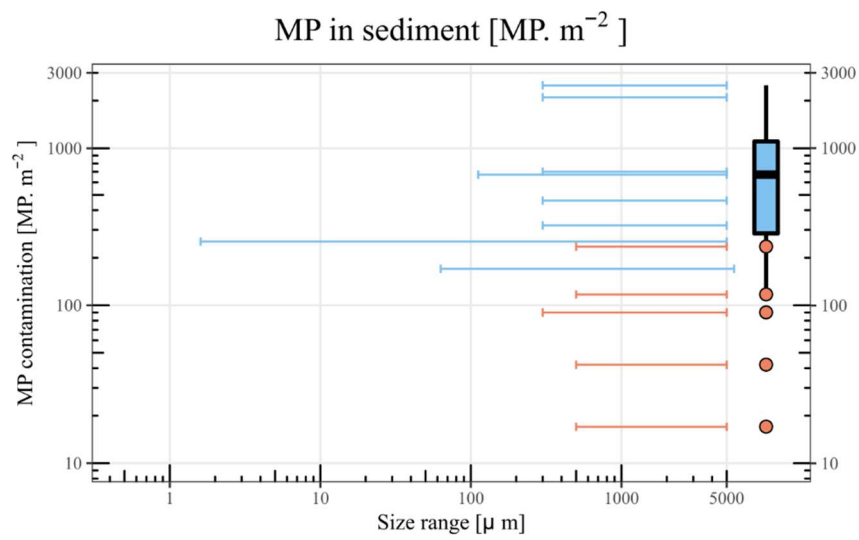
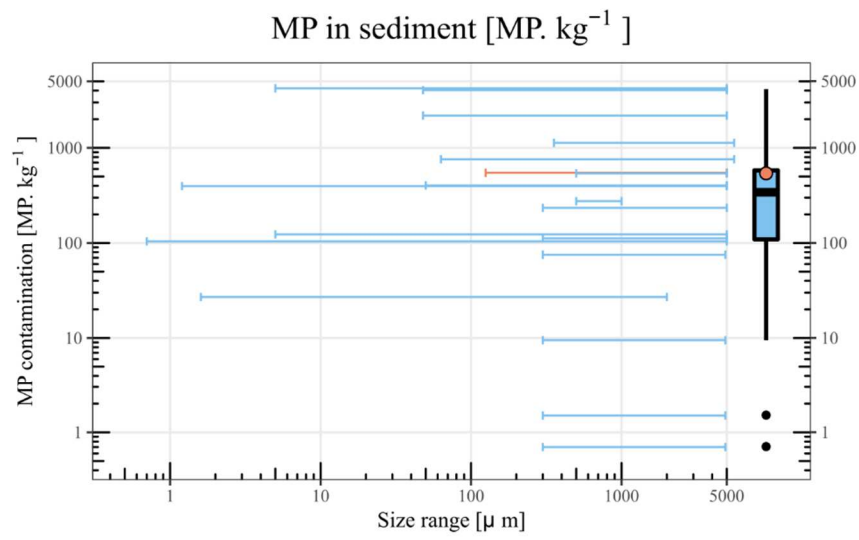
- 1198 Sci. Total Environ., 719 (2020), pp. 137512, 10.1016/j.scitotenv.2020.137512
 1199
- 1200 Wright et al., 2020 S.L. Wright, J. Ulke, A. Font, K.L.A. Chan, F.J. Kelly
 1201 **Atmospheric microplastic deposition in an urban environment and an evaluation**
 1202 **of transport**
 1203 Environ. Int., 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



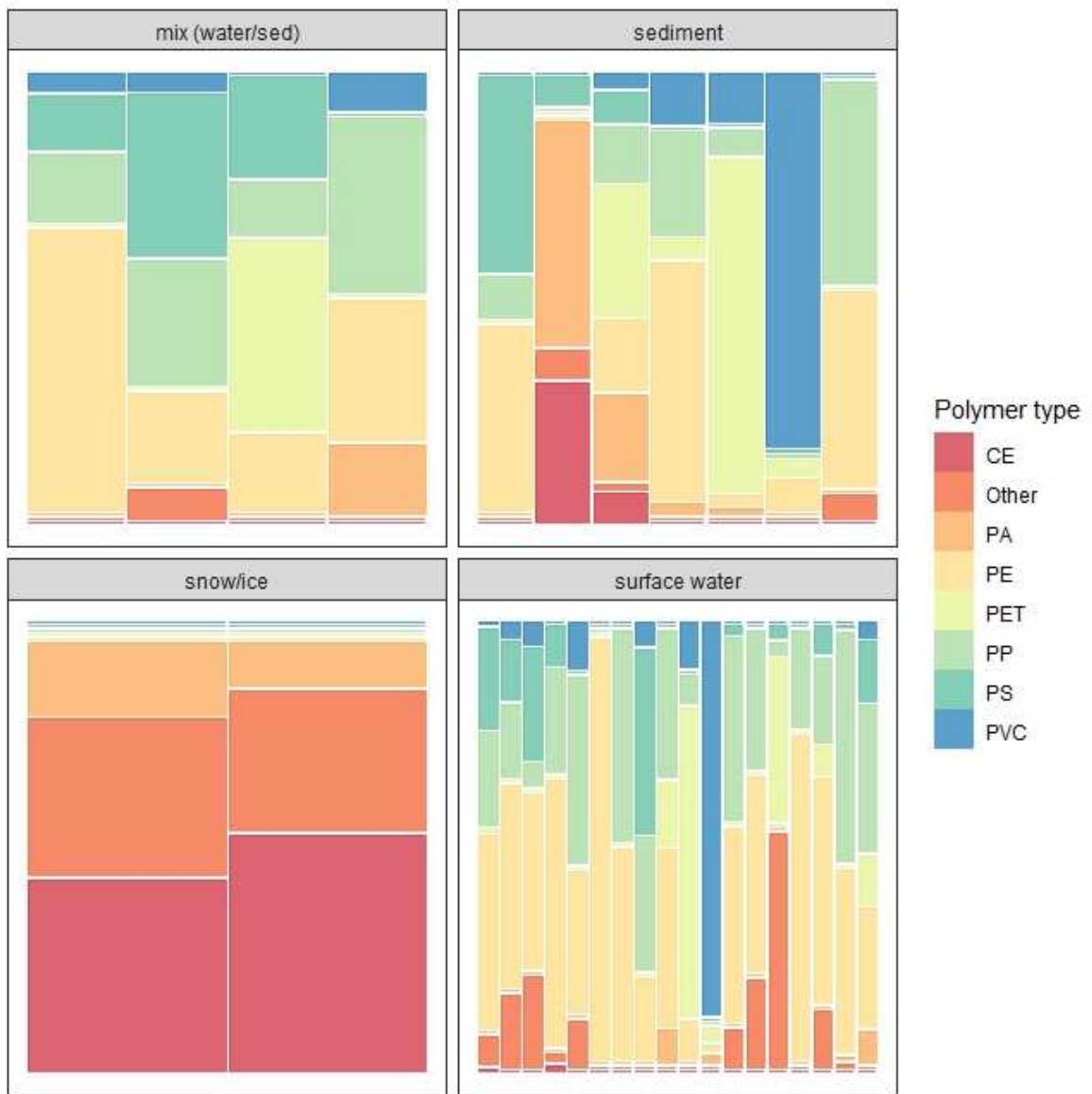
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2 Fig. 1. Microplastic concentrations on surface waters of worldwide lakes (MP/m³ or MP/km²) according to size
 3 range studied and anthropogenic pressure (rural, urban)



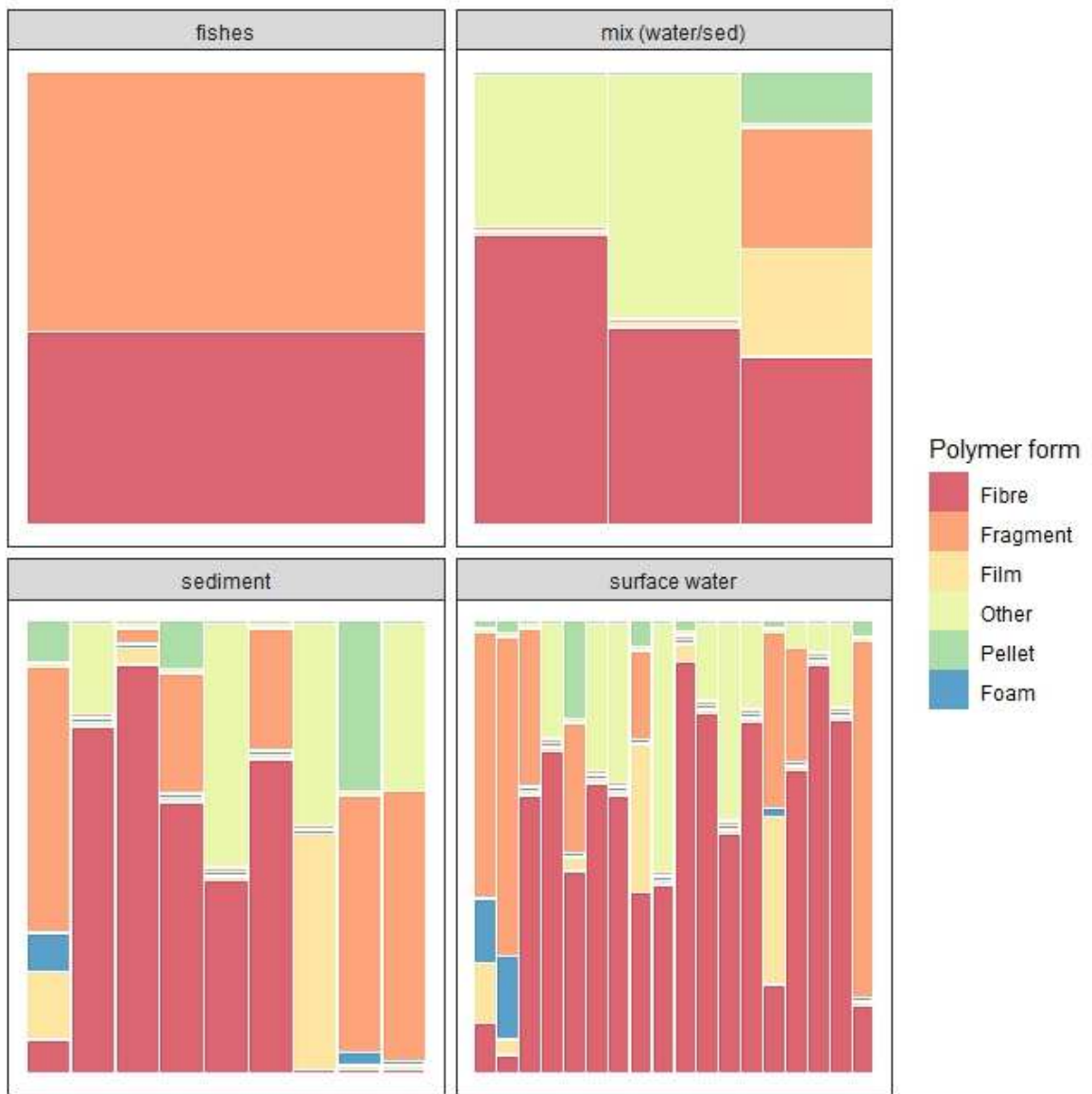
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5 Fig. 2. Microplastic concentrations in overall sediments (LBoS, LBeS, LShS) of worldwide lakes (MP/kg or
 6 MP/m²) according to size range studied and anthropogenic pressure (rural, urban)



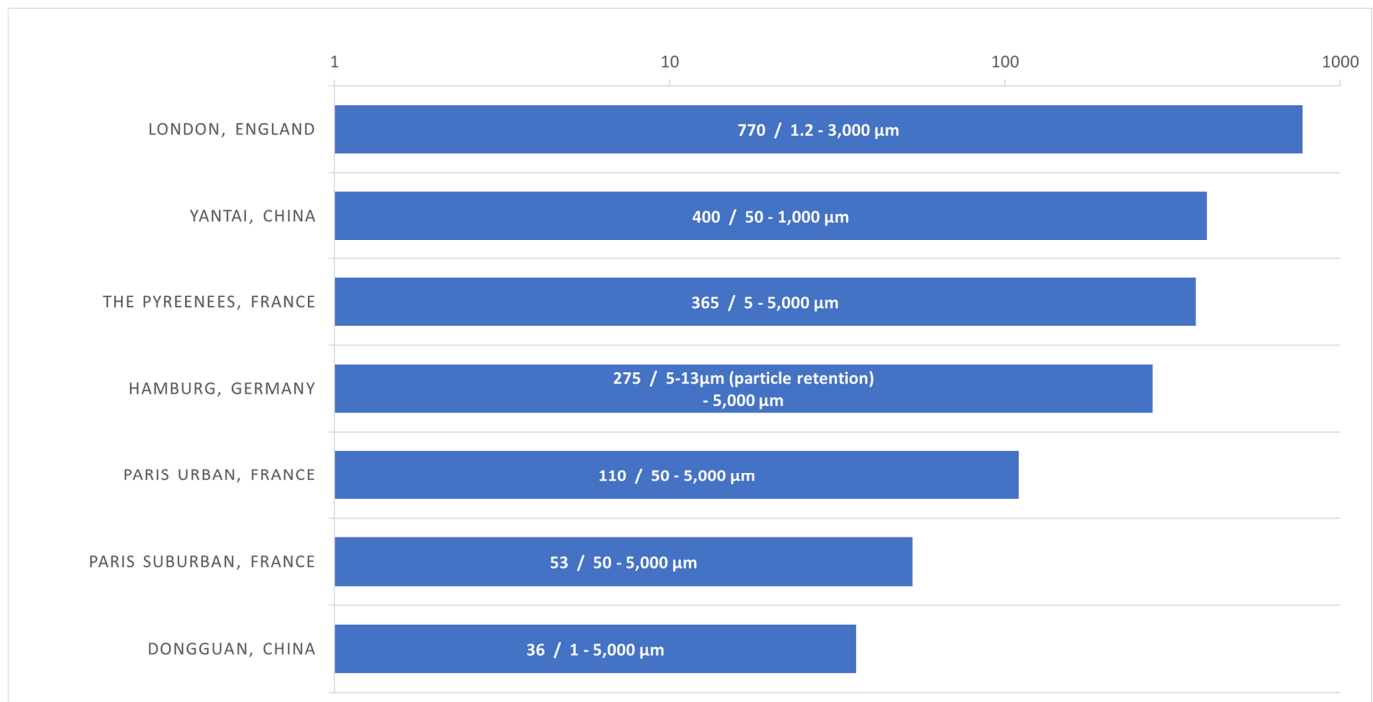
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8 Fig. 3. Distribution of the polymer types in all the compartments of the lacustrine environments across the globe



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



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12 Fig. 5. Atmospheric fallout (wet and dry) of microplastics between different geographical localization
 13 (particle/km²/day x 10⁶). For each localization, mean concentration is written and also the size class studied
 14 (μm). Numbers are rounded to the nearest unit.

Country	Mesh (μm)	Size (μm)	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 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 ± 0.18 and $1.6 \times 10^2 \pm 73$; 12 ± 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	2.3×10^5 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

