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Plastic debris exposure and effects in rivers: Boundaries for efficient ecological risk assessment

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1 **Title:** Plastic debris exposure and effects in rivers: boundaries for efficient ecological risk
2 assessment

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13
14 **Keywords:** Plastic pollution; risk management; river, ecotoxicology

15
16 **Abstract:**

17 Until recently, plastic pollution research was focused on the marine environments, and
18 attention was given to terrestrial and freshwater environments latter. This discussion paper
19 aims to put forward crucial questions on issues that limit our ability to conduct reliable plastic
20 ecological risk assessments in rivers. Previous studies highlighted the widespread presence of
21 plastics in rivers, but the sources and levels of exposure remained matters of debate. Field
22 measurements have been carried out on the concentration and composition of plastics in
23 rivers, but greater homogeneity in the choice of plastic sizes, particularly for microplastics by
24 following the recent ISO international standard nomenclature, is needed for better comparison
25 between studies. The development of additional relevant sampling strategies that are suited to
26 the specific characteristics of riverine environments is also needed. Similarly, we encourage
27 the systematic real-time monitoring of environmental conditions (e.g., topology of the
28 sampling section of the river, hydrology, volumetric flux and velocity, suspended matters
29 concentration, ...) to better understand the origin of variability in plastic concentrations in
30 rivers. Furthermore, ingestion of microplastics by freshwater organisms has been
31 demonstrated under laboratory conditions, but the long-term effects of continuous
32 microplastic exposure in organisms are less well understood. This discussion paper
33 encourages an integrative view of the issues involved in assessing plastic exposure and its

34 effects on biota, in order to improve our ability to carry out relevant ecological risk
35 assessments in river environments.

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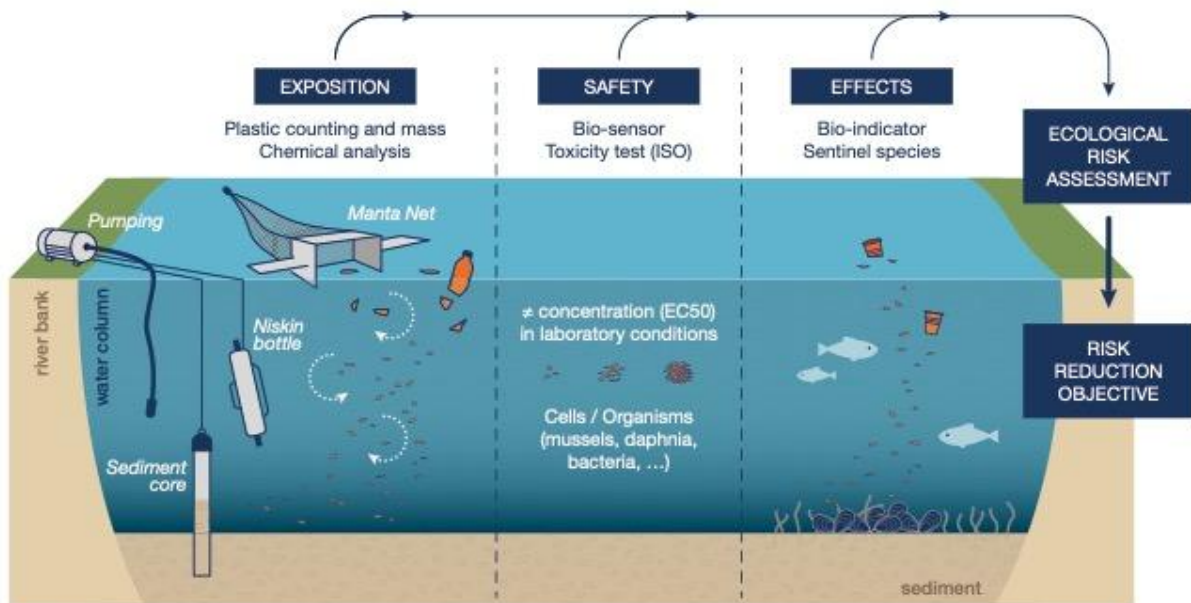
38 **Highlights:**

- 39 • Improvement in plastic sampling strategies for rivers is needed for efficient exposure
40 estimation.
- 41 • Advances in safety and effect characterization are in progress but are still insufficient.
- 42 • Links between plastic river exposure and effects are required for relevant risk
43 assessment.

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46 **Graphical abstract:**

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51 **1. Introduction**

52 For thousands of years, rivers have provided a wide range of services to human
53 civilizations. They supply cities with drinking water and serve as navigation routes. Rivers are
54 also a source of energy, both for machines (in buildings such as mills) and also for industries
55 and hydroelectric plants. Rivers are also receiving bodies for effluent and wastewater and are

56 consequently polluted by human activities, from a biological, physical or chemical point of
57 view.

58 The issue of plastic pollution is an emerging concern in rivers, as it has the potential to
59 negatively affect ecosystems, put aquatic species at risk, and result in economic losses (
60 Kurniawan et al. 2023). Although plastic collection and recycling rates have increased over
61 time, approximately 79% of all plastics ever produced have found their way into landfills or
62 the natural environment (Geyer et al. 2017). Several studies suggest that a significant portion
63 of the plastic waste found in the oceans comes from land and is carried into the oceans by
64 rivers (Meijer et al. 2021, Stokal et al. 2023). From this point of view, rivers are not only a
65 transfer route for plastics but also an environmental matrix in which plastic pollution is highly
66 concentrated compared to marine environments. However, there has been less research on
67 plastic pollution in rivers than in the ocean. This highlights the need to increase our
68 understanding of plastic pollution in freshwater ecosystems for efficient ecological risk
69 assessment and human health safety.

70 Many pollutants are present in rivers, and plastics are among these; however, awareness
71 of plastic pollution only emerged in the early 2010s (Moore et al. 2011). Field studies are now
72 available in some parts of Europe, North America and Asia (Han et al. 2023; Van Emmerik et
73 al. 2019; De Faria et al. 2021). In parallel, modeling approaches were developed to estimate
74 the amount of plastic waste entering the oceans (Meijer et al. 2021, Stokal et al. 2023). In
75 recent years, several re-evaluations of the quantities of plastic transported by rivers to the sea
76 have conducted, which estimated that approximately 500,000 tons of plastic are transported
77 by rivers (Weiss et al. 2021, Kaandrop et al. 2023). Other scientific investigations have
78 suggested the danger of plastic debris to aquatic ecosystems, with potential consequences for
79 human health. Plastic ingestion by freshwater fauna is widespread, with ingestion rates of up
80 to 33% in the Goiana River, Brazil (Possatto et al. 2011), and 13% for birds and fish in
81 French or Swiss waters (Faure et al. 2015). In particular, microplastics (MPs) have great
82 potential to enter aquatic food webs at low trophic levels and accumulate in carnivorous
83 predators via indirect ingestion of prey (Yildiz et al. 2022). MPs uptake has also been shown
84 to occur directly in vertebrates, such as in several fish species (Collard et al. 2019). Data on
85 plastic entanglement, chemical leaching and accumulation in river biota are currently lacking
86 although such data are already available for oceanic ecosystems (Høiberg et al., 2022). The
87 number of studies remains insufficient and uncertain the reliability of ecological risk assessment
88 of plastic debris in rivers.

89 A decade of research on this issue brought about awareness of plastic pollution in
90 freshwater environments, and now the scientific community has set about answering a series
91 of relevant questions. With this discussion paper, we aim to put forward crucial questions on
92 issues that limit our ability to conduct reliable plastic ecological risk assessments in rivers.
93 What are the sources of plastic in rivers? Are rivers just a source for plastic transport to the
94 oceans or are they also a sink? If plastic is stored in sediments or on riverbanks, how long
95 does this immobilization last and what are the remobilization mechanisms? What is the
96 longitudinal and vertical distribution of macro, meso-, micro- and nanoplastics in rivers? How
97 can we evaluate the risks and effects of plastics on the health of freshwater ecosystems? Do
98 we evaluate the impact of MPs on humans from a single health perspective? How strongly do
99 plastics impact the socioeconomic and cultural services provided by rivers? Even if all these
100 questions cannot be fully answered, they can lead international or national guidelines for
101 assessing risks associated with exposure to aquatic plastic pollution. This could lead to the
102 implementation of measures to reduce plastic input from identified sources. Because plastic
103 pollution monitoring methods are not robust, the answers to these questions are sometimes
104 uncertain. As a result, the implementation of standards and legislation is slow.

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106 **2. Boundaries for efficient plastic exposure assessment in rivers**

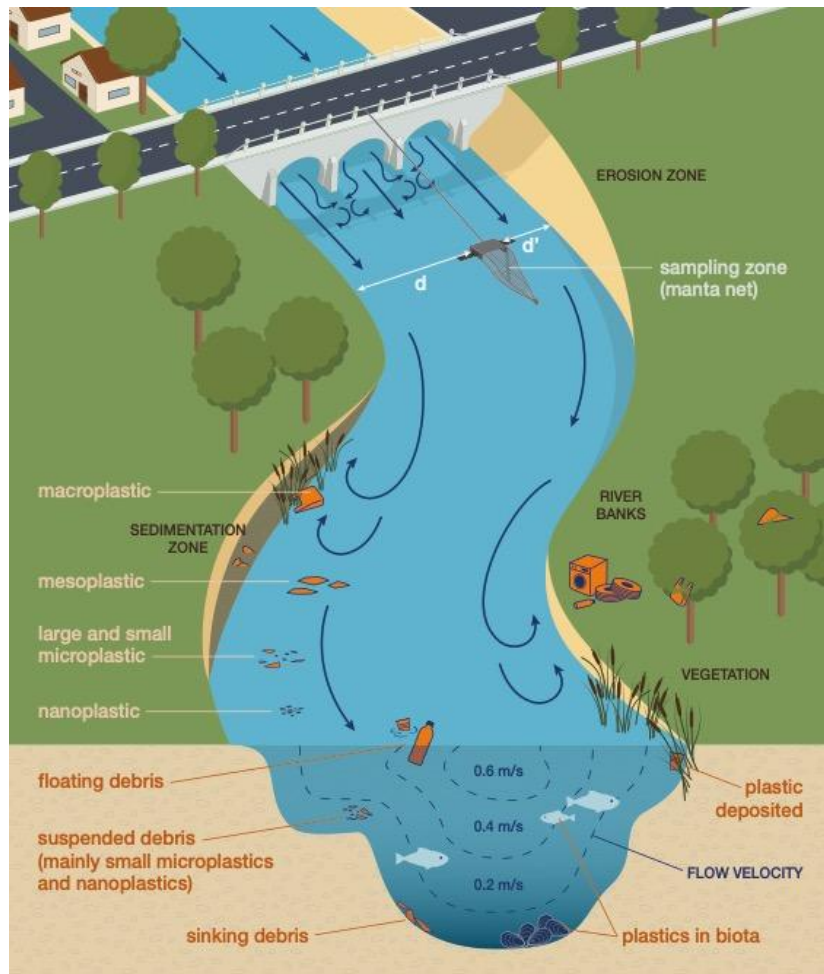
107 **2.1. Sampling strategies and methodologies**

108 Recently, plastic sampling methods have been compared in numerous review papers, and a
109 need for standardization has been noted (Bai et al. 2022; Bruge et al. 2020; Kataoka et al.
110 2023; Van Emmerik et al. 2019). For example, MPs sampling equipment in waters was
111 classified in three types, including direct sampling with containers (including Niskin bottles),
112 sampling with submersible pumps or by using various types of nets (including Manta net)
113 (Bai et al. 2022, Figure 1). Other sampling devices (including sediment core) are used in the
114 benthic environment (see Graphical abstract). The wide disparities in the sampling equipment,
115 but also sampling strategies are particularly critical for compiling data across studies, for
116 example, to better estimate riverine fluxes globally (Lofty et al. 2023; Weiss et al. 2021). It is
117 important to note that plastic monitoring is strongly impacted by river characteristics such as
118 the hydrology and topography of riverbanks and beds (Owowenu et al. 2023). In a recent
119 review paper, authors mentioned that the criteria for sampling site selection as well as
120 information on topography were often lacking (Bai et al. 2022), although the hydrodynamics
121 of sites have a major influence on the transport of plastic debris, particularly on the vertical
122 distribution of plastics. This directly impacts most monitoring methods, as sampling is

123 performed on a small proportion of the river cross-section and is most often localized at the
124 surface, but the MPs distribution is highly inhomogeneous in this section (Figure 1).
125 Hydrological conditions also play an important role in the fate and transport of MPs by
126 affecting the distribution of MPs in cross-sections of rivers, their distribution between water
127 and sediment, and the mobilization of sediment (and hence of sedimented plastic debris)
128 (Hurley et al. 2018). The monitoring of sediments is even more complicated than the
129 monitoring of river water, as sedimentation of MPs is closely linked to a river's topography
130 and hydrology. These factors contribute to areas where MPs accumulate, as well as to the
131 remobilization of these particles from sediments and their downstream transport in the river
132 (Liro et al. 2020).

133 In a recently published review article, a sampling mode that covers the entire cross section of
134 a river was proposed (Bai et al. 2022). This type of sampling imposes severe material
135 constraints on sites. As an alternative, the authors proposed profile monitoring with
136 measurements in the vertical and horizontal directions of a river cross-section (Figure 1). This
137 original sampling strategy requires substantial effort in terms of sampling and analyses, but it
138 will undoubtedly bring to light the answers to many questions on the spatial variations of
139 plastics observed to date. Long-term temporal monitoring strategy of riverine plastic pollution
140 are also needed, by considering various flow regimes (measured at least under base and high
141 flow regimes) that may greatly affect the trend of plastic concentration in the river. To
142 investigate the variation characteristics of riverine plastic debris loads, the volumetric flux (in
143 $\text{m}^3.\text{s}^{-1}$) and velocity (in $\text{m}.\text{s}^{-1}$) are crucial measurements in order to draw plastic distribution
144 curve under the specific flow regime (Figure 1). We also encourage the measurement of other
145 real-time data during sampling - such as temperature, salinity, turbidity, suspended matter and
146 sediment concentration – in order to look for correlations among data with regard to plastic
147 concentration. The characteristics of the sampling section of the river must also be
148 documented, such as the presence of vegetation, river curvature (erosion zone or
149 sedimentation zone) and depth, as well as the presence of man-made structures, such as dams,
150 hydroelectric power stations, bridges or artificial banks, which may have an impact on river
151 hydrodynamics and on plastic transport (Ita-Nagy et al. 2022) (Figure 1).

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Figure 1. Illustration of the fate of various plastic items in rivers according to their characteristics (size and buoyancy) and flow velocity. Sampling a single spot in the cross section of the river, leads to great uncertainties in estimating the plastic fluxes of rivers. Multiple sampling points in transverse sections of rivers may be necessary to represent the heterogeneous distribution of plastics in rivers. A better description of river topography and hydrology at the sampling points would also facilitate comparisons between studies. Velocity flow data, which vary with depth, are given for information only.

162 2.2. Microplastic categorization into size ranges

163 While the upper size limit for categorizing and sampling MPs is universally applied and set to
164 5 mm, the lower limit varies widely from one study to another. This topic has been the subject
165 of much discussion, and there is a growing awareness of the need for standardization to
166 enable comparisons between studies (Razeghi et al. 2022; Weiss et al. 2021). As the detection
167 limit decreases, two marked phenomena have been commonly observed. First, the lower the
168 limit is, the higher the concentrations expressed in particles m^{-3} . Second, the proportion of
169 fibers increases as the lower limit decreases (Cordova et al. 2022; Weiss et al. 2021). The
170 study of the multidimensionality of MPs has enabled the establishment of probability density
171 functions specific for different aquatic compartments (Koelmans et al. 2020; Kooi et al.

172 2021). These methods enable us to mathematically correct the concentrations obtained with
173 distinct size limits, allowing intercomparison.

174 As MPs quantification studies address very distinct size classes due to the detection limit and
175 technical constraints inherent in every study, we propose that subcategories are adopted and
176 that a universal consensus is reached to define MP size ranges. Legislation by several
177 countries (e.g., US, Canada, UK, France, Italy, Belgium, Sweden, Australia, and New
178 Zealand) typically sets 5 mm as an upper size limit for MPs. We recommend to follow the
179 recent ISO international standard nomenclature in which MPs would be classified in one of
180 two size groups: 1 to 1000 μm and 1000 to 5000 μm in any dimension (International
181 Organization for Standardization, 2020). These two fractions are typically characterized by
182 distinct techniques, the larger fraction by attenuated total reflectance Fourier transformed
183 infrared spectroscopy and the smaller fraction by micro spectroscopy (infrared or Raman).
184 Hence, depending on the detection limit of the method used, subcategories can be addressed
185 and compiled for the 1 to 1000 μm fraction. In addition to the cutoff limit for sampling,
186 subsamples can be obtained by cascade filtration, for example (Bannick et al. 2019). The
187 implementation of cascade filtration requires careful management of filter clogging, as has
188 been discussed with respect to the clogging of systems during sampling.

189

190 **2.3. Importance of polymer composition and density on plastic spatial distribution**

191 A review revealed that the most common polymer type detected in rivers is polyethylene (PE)
192 (42%), followed by polypropylene (PP) (30%) and polystyrene (PS) (11%) (Bai et al. 2022).
193 Other common polymer types, such as poly(ethylene terephthalate) (PET), polyamide, and
194 polyester, have also often been observed (Bai et al. 2022). Polymer type variation from one
195 river to another, seasonal fluctuations or differences observed along rivers were observed.
196 Although PE and PP (commonly used in disposable plastic products) made up the majority of
197 polymers in the rivers studied, PS foam (widely used in food packaging and impact-resistant
198 containers) was found to be the most abundant in Hong Kong waters (Cheung et al. 2018) or
199 for a very large proportion together with polyolefin in Saigon River (van Emmerick et al.
200 2018). Despite the uncertainties generated by sampling, certain types of sources have been
201 identified for primary MPs, whereas the origin of secondary MPs is more uncertain. For
202 example, textile fibers are thought to originate from wastewater treatment plants, and tire
203 particles originate from rainwater runoff (Arias et al. 2022).

204 As analytical methods became more sensitive, new types of polymers are emerging. The paint
205 particles that contain toxic biocides were initially studied in the marine environments and

206 have also been also detected in rivers (De-la-Torre et al. 2023). MPs from tyre wear could be
207 major contributors to particulate pollution that comes to rivers mainly from stormwater runoff
208 (Miera-Domínguez et al. 2024). There is also increasing concern about fluorinated polymers
209 in rivers, which is assumed to have a higher potential toxicity than conventional polymers
210 (PE, PP and PS) (Lohmann et al. 2020).

211 After several years of investigation into estimating the extent of MPs sedimentation in the
212 oceans, the scientific community has reached a consensus that marine sediments are sinks for
213 MPs (Kane et al. 2020). An increasing number of studies are reporting very high
214 concentrations of MPs in sediments (Bai et al. 2022; Claessens et al. 2013; Kumar et al. 2021;
215 Scherer et al. 2020a), thus suggesting that river sedimentation of MPs is also a major process
216 in rivers. However, there are significant hydrological variations in rivers with meteorology,
217 particularly when heavy rainfall occurs, which means that rivers are very different from
218 oceans and that remobilization of MPs from sediments and downstream transport or transport
219 to oceans is possible. We recommend extending studies on the MPs remobilization from river
220 sediments together with its relation to the river's topography and hydrology. This will require
221 a decompartmentalization of studies, which up to now have focused on either sediments or
222 river waters, in order to gain a better understanding of the interactions between MPs and biota
223 in both benthic and pelagic ecosystems.

224

225 **3. Boundaries for assessing the toxicity of microplastics in rivers**

226 The development and standardization of toxicity and ecotoxicity studies are essential for
227 ecological risk assessment to provide information beyond the temporal and spatial dynamics
228 of exposition to plastic debris (see Graphical abstract). To date, most studies have focused on
229 safety measurements (mainly toxicity tests or tests using biosensors) but not on *in situ* effects
230 on sentinel species or complex natural communities (e.g. bioindicators) (see Graphical
231 abstract). Standardized methods have been used to indicate toxic effects at the individual
232 level, for example, by exposing the crustacean *Daphnia magna* (Martins et al. 2018; Xu et al.
233 2020), the bivalve *Dreissena polymorpha* (Weber et al. 2020), the crustacean *Hyaella azteca*
234 (Au et al. 2015), the dipteran *Chironomus riparius* (Scherer et al. 2020b) or the zebrafish
235 *Danio rerio* (Karami et al. 2017) to micro- and nanoplastics. The use of bacterial biosensors
236 has also made it possible to offer a simple, environmentally-friendly alternative to standard
237 analysis techniques (Popenda et al. 2024). These studies, which were conducted in
238 laboratories, employed simplified exposure conditions, such as single polymer types, single
239 particle sizes, sphere-shaped particles, or high particle concentrations, which may not

240 accurately reflect real-world environmental conditions. Suggestions to improve the relevance
241 of plastic toxicity studies and standards are as follows.

242

243 **3.1. Consideration of the concentration, size, shape, biofilm formation potential and** 244 **chemical composition of plastics in toxicity tests**

245 Toxicity tests are generally unable to accurately represent environmental plastic
246 concentrations, as these concentrations vary depending on factors such as location,
247 meteorological conditions, and time. The concentration of MPs used in toxicological studies
248 generally ranges between 20 and 2,000 mg/L (Scherer et al. 2017), which is several orders of
249 magnitude higher than the highest concentration recovered in riverine environments (Weiss et
250 al. 2021).

251 Moreover, the majority of MPs that have been tested are small MPs, ranging between 1
252 and 100 μm , which do not represent the large diversity of plastic sizes found in the
253 environment. We suggest addressing the knowledge gap regarding nanoplastics in particular,
254 as they may represent the most prevalent form of plastics in terms of particle numbers in the
255 environment; additionally, they have a high surface-to-volume ratio, which is likely to favor
256 micropollutant sorption (Yu et al. 2021). A review paper pointed out counterintuitive results,
257 revealing that a decrease in particle size did not result in an increase in toxicity (Jones et al.
258 2019), but further studies are needed for particular nanoplastics.

259 The shape of plastic particles and their colonization by natural biofilms may also have
260 an impact on the environmental representativeness of toxicity results. Most of related studies
261 employed pristine primary MPs that were intentionally manufactured as small particles with
262 uniform spherical or cylindrical shapes, which represent a negligible part of the total MP
263 pollution worldwide. Irregular fragments have been shown to induce greater toxicity than
264 regular-shaped fragments on *Daphnia magna* (Frydkjær et al. 2017). The ubiquitous presence
265 of microbial biofilms on plastics influences their buoyancy and palatability (Jacquin et al.
266 2019), but only a few toxicological studies have employed a preincubation step on plastic
267 pieces (Lear et al. 2021). According to the model species used, what it preferentially ingests
268 and the experimental goal, we recommend the use of irregular fragments that are preincubated
269 for at least one month in a natural river to allow mature biofilm formation (Odobel et al.
270 2021) for more realistic experimental conditions.

271

272 **3.2. Chemicals associated with plastics and potential toxicity**

273 Over 13,000 chemical substances are associated with plastic production, and among
274 these, more than 3,000 are of potential concern due to their hazardous properties (UNEP
275 2023). Additives can constitute up to 80% of the total weight of polyvinyl chloride (PVC) and
276 generally contain 93% polymer resin and 7% additives on average by mass, consisting of
277 stabilizers, pro-oxidants, surfactants, inorganic fillers or pigments (Geyer et al. 2017).
278 Comparisons between studies on the leaching of additives in laboratory analyses are hindered
279 by methodological differences, such as variations in the leaching period, initial state of
280 plastics, temperature, or presence of light. Even though life cycle assessment covers the toxic
281 impacts of several thousands of chemicals, models to assess the toxic impacts of plastic
282 additives are only emerging (Casagrande et al. 2024).

283 Moreover, determining the exact composition of combined polymers and additives is
284 often difficult, making comprehensive analysis of leachates challenging (Gunaalan et al.
285 2020). Standardization of leaching procedures (e.g., leaching time, T°C, agitation speed, light,
286 shape and oxidation state) is impeded by a lack of knowledge of leaching processes from
287 plastics under environmental conditions. Clear labeling and listing of plastic additive content
288 would greatly facilitate the establishment of relevant strategies for ecological risk assessment.

289 Another limitation in toxicity tests is the discrepancy between the polymer types
290 utilized in these studies and their actual presence in the environment. PVC and PS are
291 extensively used in toxicity tests because they are available as standardized microbeads on the
292 market; however, their presence in riverine environments is low compared to other types of
293 plastics. PP is rarely used, whereas it is the second most abundant polymer on river surfaces
294 after PE (Fan et al. 2019).

295

296

3.3. Ecotoxicity of plastics

297 Assessing the impact of plastics on organisms in their natural environment is difficult
298 because riverine environments are already affected by various chemicals and wastes, and
299 other factors, such as extreme events due to global warming, habitat degradation, and
300 diseases, can also cause stress (Horton et al. 2017). As a result, the ecotoxicity observed in
301 organisms cannot be attributed solely to plastics, even if they are found in the organisms
302 (Leistenschneider et al. 2023).

303 Experiments conducted under controlled mesocosm conditions may be useful for
304 mimicking the impact of plastics on biodiversity and ecosystem functions, but such
305 experiments are limited. For example, elevated MPs concentrations had only a slight impact
306 on the population dynamics of most taxa in freshwater food webs in mesocosm experiments,

307 despite the propensity of MPs to be directly or indirectly transferred to higher trophic levels
308 (Yildiz et al. 2022). Such experiments are rare, and further studies are needed to fill the
309 substantial knowledge gap between single-species laboratory experiments and community-
310 level studies of plastics in freshwater habitats. In particular, further long-term in-situ
311 freshwater mesocosms experiment with complex food web structure should be encouraged to
312 fully understand potential threats of MPs to biodiversity and ecosystem functioning in rivers.

313

314 **4. Conclusions**

315 Plastic litter is now a criterion for water quality assessment in several countries and is
316 included in the Canadian Water Quality Guidelines for the Protection of Aquatic Life (1999),
317 the European amendment in 2019 to the Marine Strategy Framework Directive and the United
318 States amendment “Beaches Environmental Assessment and Coastal Health Act” (2000).
319 None of these guidelines set concentration thresholds, and they focus mainly on
320 macroplastics, not on meso- or microplastics, even though the latter have received major
321 attention in scientific literature in the last decade. Numerous studies on the standardization of
322 sampling and analysis methods for assessing plastic exposure in rivers have been performed,
323 which have enabled the scientific community to highlight practices that could be improved.
324 However, this kind of approach should not prevent scientists from "thinking outside the box"
325 or proposing methods that break with the majority of practices adopted to date, offering new
326 alternatives. For example, the much higher mass concentration of small microplastics (SMPs,
327 25 to 500 μm) to the large microplastics (LMPs, 500 μm to 5 mm) with SMP/LMP ratios up
328 to 1000 in some rivers (Landebrit et al. 2024) has direct implication on toxicological studies
329 that should take into account the different stages of plastic fragmentation to LMP, SMP and
330 nanoplastics. Because plastic debris has a multitude of chemical compositions and physical
331 properties and exists in a size continuum, monitoring is complex, and the strong impact of
332 hydrological, topographical and meteorological conditions on measured concentrations makes
333 it difficult to establish transport or transformation mechanisms in rivers. This has delayed the
334 introduction of regulations and legislation to limit plastic pollution. However, the numerous
335 studies showing the serious ecological and public health consequences of this type of
336 pollution suggest that the rapid introduction of restrictive measures is necessary.

337 Over the past decade, international efforts, laws, and policies addressing the issue of
338 plastic pollution in the environment have noticeably increased. This growing concern has
339 been reflected in various initiatives, including those led by the G20, G7, and UNEA, which
340 have all supported the establishment of an international treaty currently being negotiated, the

341 UNEP Resolution 5/14 (2022). National or international restrictions on the marketing of
342 single-use plastics (including bags, cotton bud sticks, cutlery, plates, straws, stirrers, cups,
343 ring carriers, beverage containers made of expanded polystyrene, exfoliating rinse-off
344 cosmetic products, and all products made of oxo-degradable plastics) set a precedent,
345 contributing to growing media coverage and public awareness. None of these initiatives were
346 based on relevant evaluations of ecological risk due to the challenges involved in testing the
347 wide range of plastics of various compositions found in the targeted plastic items. There is an
348 urgent need to include quantification of the potential impact of plastic leakage in life-cycle
349 assessment methods, to better reflect the risks that plastic emissions pose to the quality of
350 river ecosystems (Corella-Puertas et al. 2023). By developing and analyzing extensive
351 datasets on plastic exposure in different organisms and environments, we can progress from
352 basic monitoring to conducting comprehensive ecological risk assessments of plastic pollution
353 in riverine ecosystems. Achieving these objectives is crucial as we strive for a sustainable
354 future in terms of human and environmental health.

355

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359

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569

570 **Declarations**

571

572 **-Ethical Approval:** This article follows the Committee on Publication Ethics (COPE)
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