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

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Keywords: Plastic pollution; risk management; river, ecotoxicology

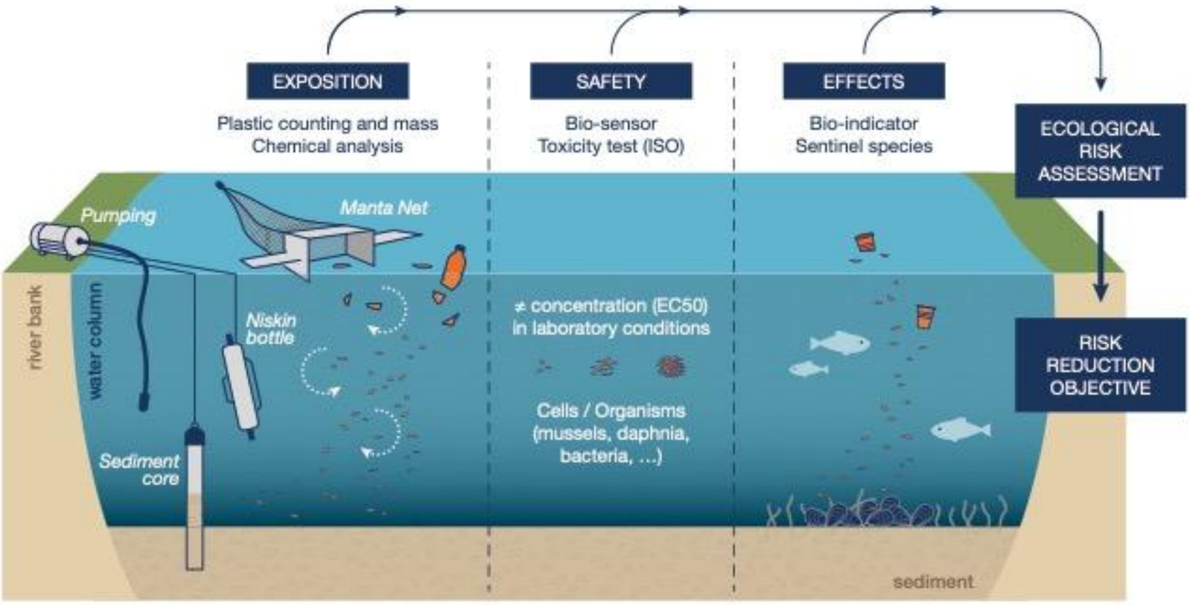
Abstract:
Until recently, plastic pollution research was focused on the marine environments, and attention was given to terrestrial and freshwater environments latter. This discussion paper aims to put forward crucial questions on issues that limit our ability to conduct reliable plastic ecological risk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

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

Highlights:

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

Graphical abstract:



1. Introduction

For thousands of years, rivers have provided a wide range of services to human civilizations. They supply cities with drinking water and serve as navigation routes. Rivers are also a source of energy, both for machines (in buildings such as mills) and also for industries and hydroelectric plants. Rivers are also receiving bodies for effluent and wastewater and are consequently polluted by human activities, from a biological, physical or chemical point of view.

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

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

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

2. Boundaries for efficient plastic exposure assessment in rivers

2.1. Sampling strategies and methodologies

Recently, plastic sampling methods have been compared in numerous review papers, and a need for standardization has been noted (Bai et al. 2022; Bruge et al. 2020; Kataoka et al. 2023; Van Emmerik et al. 2019). For example, MPs sampling equipment in waters was classified in three types, including direct sampling with containers (including Niskin bottles), sampling with submersible pumps or by using various types of nets (including Manta net) (Bai et al. 2022, Figure 1). Other sampling devices (including sediment core) are used in the benthic environment (see Graphical abstract). The wide disparities in the sampling equipment, but also sampling strategies are particularly critical for compiling data across studies, for example, to better estimate riverine fluxes globally (Lofty et al. 2023; Weiss et al. 2021). It is important to note that plastic monitoring is strongly impacted by river characteristics such as the hydrology and topography of riverbanks and beds (Owowenu et al. 2023). In a recent review paper, authors mentioned that the criteria for sampling site selection as well as information on topography were often lacking (Bai et al. 2022), although the hydrodynamics of sites have a major influence on the transport of plastic debris, particularly on the vertical distribution of plastics. This directly impacts most monitoring methods, as sampling is performed on a small proportion of the river cross-section and is most often localized at the surface, but the MPs distribution is highly inhomogeneous in this section (Figure 1). Hydrological conditions also play an important role in the fate and transport of MPs by affecting the distribution of MPs in cross-sections of rivers, their distribution between water and sediment, and the mobilization of sediment (and hence of sedimented plastic debris) (Hurley et al. 2018). The monitoring of sediments is even more complicated than the monitoring of river water, as sedimentation of MPs is closely linked to a river's topography and hydrology. These factors contribute to areas where MPs accumulate, as well as to the remobilization of these particles from sediments and their downstream transport in the river (Liro et al. 2020).

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

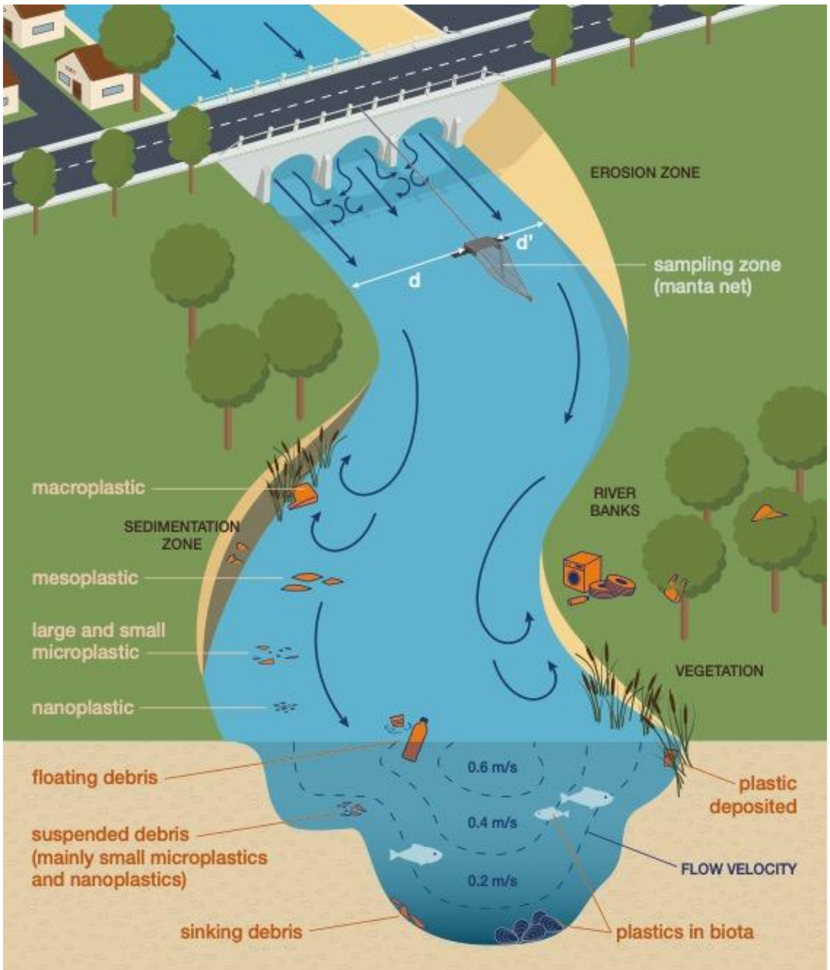


Figure 1. Illustration of the fate of various plastic items in rivers according to their characteristics (size and buoyancy) and flow velocity. Sampling a single spot in the cross section of the river, leads to great uncertainties in estimating the plastic fluxes of rivers. Multiple sampling points in 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.

2.2. Microplastic categorization into size ranges

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

2021). These methods enable us to mathematically correct the concentrations obtained with distinct size limits, allowing intercomparison.
As MPs quantification studies address very distinct size classes due to the detection limit and technical constraints inherent in every study, we propose that subcategories are adopted and that a universal consensus is reached to define MP size ranges. Legislation by several countries (e.g., US, Canada, UK, France, Italy, Belgium, Sweden, Australia, and New Zealand) typically sets 5 mm as an upper size limit for MPs. We recommend to follow the recent ISO international standard nomenclature in which MPs would be classified in one of two size groups: 1 to 1000 μm and 1000 to 5000 μm in any dimension (International Organization for Standardization, 2020). These two fractions are typically characterized by distinct techniques, the larger fraction by attenuated total reflectance Fourier transformed infrared spectroscopy and the smaller fraction by micro spectroscopy (infrared or Raman). Hence, depending on the detection limit of the method used, subcategories can be addressed and compiled for the 1 to 1000 μm fraction. In addition to the cutoff limit for sampling, subsamples can be obtained by cascade filtration, for example (Bannick et al. 2019). The implementation of cascade filtration requires careful management of filter clogging, as has been discussed with respect to the clogging of systems during sampling.

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

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

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

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

3. Boundaries for assessing the toxicity of microplastics in rivers

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

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

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

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

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

3.2. Chemicals associated with plastics and potential toxicity

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

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

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

3.3. Ecotoxicity of plastics

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

Experiments conducted under controlled mesocosm conditions may be useful for mimicking the impact of plastics on biodiversity and ecosystem functions, but such experiments are limited. For example, elevated MPs concentrations had only a slight impact on the population dynamics of most taxa in freshwater food webs in mesocosm experiments, despite the propensity of MPs to be directly or indirectly transferred to higher trophic levels (Yildiz et al. 2022). Such experiments are rare, and further studies are needed to fill the substantial knowledge gap between single-species laboratory experiments and community-level studies of plastics in freshwater habitats. In particular, further long-term in-situ freshwater mesocosms experiment with complex food web structure should be encouraged to fully understand potential threats of MPs to biodiversity and ecosystem functioning in rivers.

4. Conclusions

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

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

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

Acknowledgements

We are grateful to Mikaël Kedzierski and Guigui PA, VF, JS, and JP for insightful comments on the manuscript.

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Declarations

-Ethical Approval: This article follows the Committee on Publication Ethics (COPE) guidelines. including the ethical responsibilities of the authors. The authors declare that they obtained study-specific approval from the appropriate ethics committee for the research content of this article.

-Consent to Participate: All the authors agreed to participate in coauthorship. The authors have no competing interests to declare that they are relevant to the content of this article.

-Consent to Publish: All the coauthors agreed with the content of this article, and they all provided explicit consent for submission. The authors obtained consent from the responsible authorities at the institute/organization where the work was carried out before the work was submitted.

-Author Contributions (CRediT taxonomy): Jean-François Ghiglione: Conceptualization, Funding acquisition, Project administration, Resources, Supervision, Visualization, Writing - original draft, review & editing; **Alexandra ter Halle:** Conceptualization, Funding acquisition, Project administration, Resources, Supervision, Visualization, Writing - original draft, review & editing.

-Funding: This work was supported by a PLASTRANSFER grant supported by the French Agency for Ecological Transition (ADEME) and the French Biodiversity Agency (OFB) and by the European Union's Horizon 2020 research and innovation project AtlantECO under grant agreement No. 862923.

-Competing Interests: The authors have no relevant financial or nonfinancial interests to disclose**.**

-Availability of data and materials: The datasets and materials used and/or analyzed in the current study are available upon reasonable request.