



**HAL**  
open science

## The missing ocean plastic sink: gone with the rivers

Lisa Weiss, Wolfgang Ludwig, Serge Heussner, Miquel Canals, Jean-François Ghiglione, Claude Estournel, Mel Constant, Philippe Kerhervé

### ► To cite this version:

Lisa Weiss, Wolfgang Ludwig, Serge Heussner, Miquel Canals, Jean-François Ghiglione, et al.. The missing ocean plastic sink: gone with the rivers. *Science*, 2021, 373 (6550), pp.107-111. 10.1126/science.abe0290 . hal-03718112

**HAL Id: hal-03718112**

**<https://hal.science/hal-03718112>**

Submitted on 8 Jun 2023

**HAL** is a multi-disciplinary open access archive for the deposit and dissemination of scientific research documents, whether they are published or not. The documents may come from teaching and research institutions in France or abroad, or from public or private research centers.

L'archive ouverte pluridisciplinaire **HAL**, est destinée au dépôt et à la diffusion de documents scientifiques de niveau recherche, publiés ou non, émanant des établissements d'enseignement et de recherche français ou étrangers, des laboratoires publics ou privés.

## **Title: The missing ocean plastic sink: gone with the rivers**

**Authors:** Lisa Weiss<sup>1\*</sup>, Wolfgang Ludwig<sup>1</sup>, Serge Heussner<sup>1</sup>, Miquel Canals<sup>2</sup>, Jean-François Ghiglione<sup>3</sup>, Claude Estournel<sup>4</sup>, Mel Constant<sup>1†</sup>, Philippe Kerhervé<sup>1</sup>

### **Affiliations:**

<sup>1</sup>CEFREM, UMR 5110 University of Perpignan - CNRS, F-66860 Perpignan Cedex, France.

<sup>2</sup>CRG Marine Geosciences, Department of Earth and Ocean Dynamics, University of Barcelona, E-08028 Barcelona, Spain.

<sup>3</sup>LOMIC, UMR 7621 Sorbonne University - CNRS, F-66650 Banyuls-sur-Mer, France.

<sup>4</sup>LEGOS, UMR 5566 University Toulouse III – CNRS/CNES/IRD/UPS, F-31400 Toulouse, France.

†Current address: IMT Lille Douai, GCE, F-59508 Douai, France.

\*Corresponding author. Email: lisa.weiss@univ-perp.fr

### **One Sentence Summary:**

**New data drastically drop previous estimates of microplastic river fluxes to the ocean, reducing the need for a missing ocean plastic sink.**

### **Abstract:**

**Plastic floating at the ocean surface, estimated at tens to hundreds of kilotons, represents only a small fraction of the estimated several million tons annually discharged by rivers. Such an imbalance promoted the search for a “missing plastic sink” that could explain a rapid removal of river-sourced plastics from the ocean surface. Based on an in-depth statistical reanalysis of updated data on microplastics – a size fraction for which both ocean and river sampling rely on equal techniques – we demonstrate that current river flux assessments are overestimated by two to three orders of magnitude. Accordingly, average residence time of microplastics at the ocean surface rises from a few days to several years, strongly reducing the need for a missing sink.**

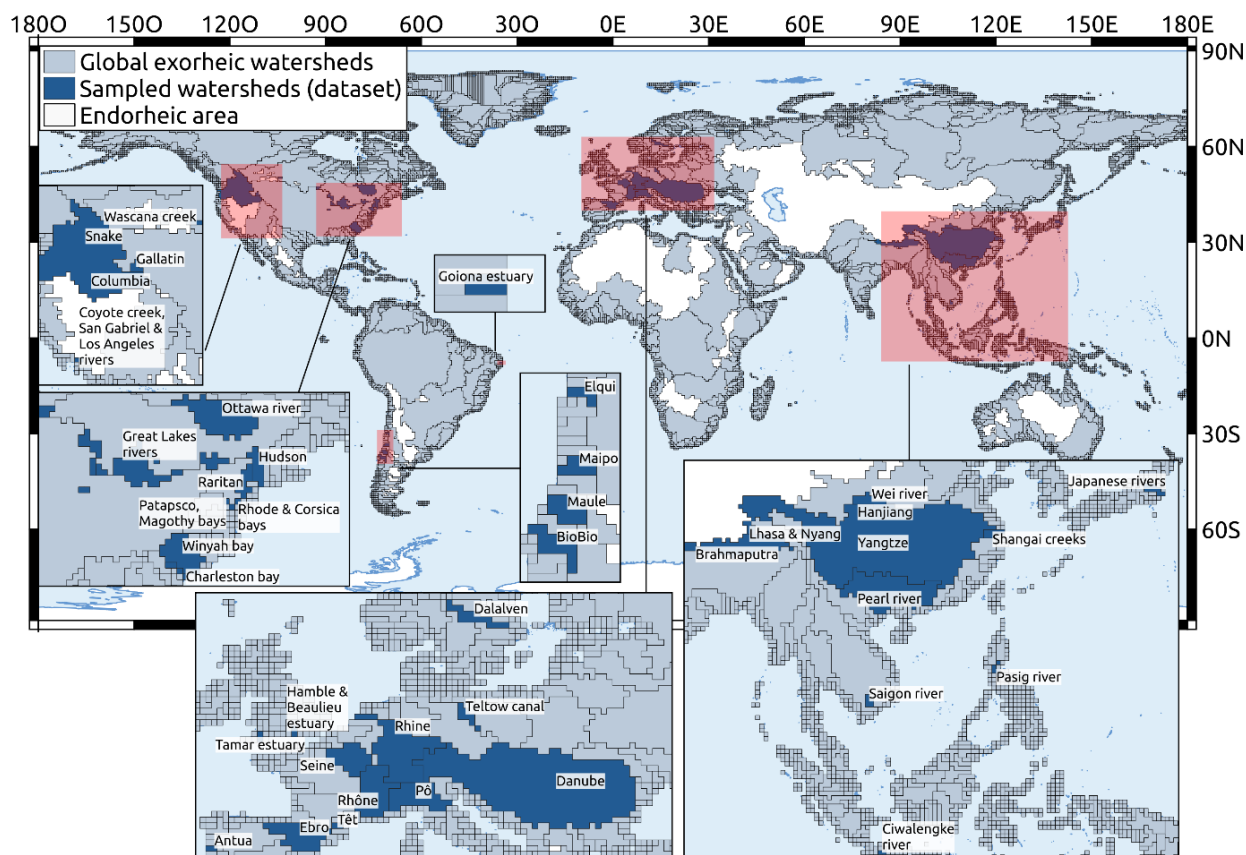
### **Main Text:**

Over recent years, plastic pollution has become a growing concern that attracted public awareness and fostered numerous studies to identify and quantify plastic waste sources, transport and fate within terrestrial, coastal and marine environments (1, 2). A central issue has been the quantification of discharge to the global ocean, considered as a major sink for plastic debris (3). Leakage from waste generation and inadequate disposal on land – the mismanaged plastic waste (MPW) – was initially identified as the main driver for plastic discharge to the ocean with a potential annual transfer of 4.8 to 12.7 Mt (4). Rivers are recognized to be the principal conveyors in this transfer (5, 6). Correlations between estimated river plastic fluxes and MPW in drainage basins confirmed, in subsequent studies, that maximum river fluxes of total plastics could reach

2.4 to 4.0 Mt yr<sup>-1</sup> (7, 8). Among the different plastic size categories generally recognized, the <5 mm microplastics (MPs hereafter) represent the size fraction for which both river flux and surface oceanic MP stock estimates rely on equal sampling techniques, thus allowing direct comparison (Supplementary Materials and Methods 3.1). Narrowing the figures to this better-documented category yields an average MP river input of 1.5 Mt yr<sup>-1</sup> (9). Furthermore, the amount of plastic floating at the ocean surface, mainly composed by MPs (10, 11), is currently estimated at 14.4 kt (12), 35.5 - 66.1 kt (13) or even 236 kt (11), depending on data availability, modeling approach and the upper particle size limit considered. So far, compelling evidence for a rapid growth of the floating plastic stock in the ocean is missing (14, 15). Therefore, dividing the relevant amount by the annual MP flux from rivers provides a conservative indication of the average residence time of MP debris at the ocean surface. A global river discharge in the range of million tons per year accordingly implies an unrealistic short residence time of MPs at the ocean surface, i.e. in the range of days to weeks.

As with the “missing carbon sink” concept (16) that fostered fundamental research on the global carbon cycle in the early 1990s, the idea of a “missing ocean plastic sink” emerged from such an overwhelming difference between the amount of MPs supposedly brought by rivers and the stocks accumulating at the ocean surface (12, 17, 18) and triggered research on the fate of plastic debris in the marine environment. Several processes have been proposed to be prime actors in preventing ever-growing plastic accumulation at the ocean surface. Among them is rapid accumulation in living biomass (19), quick loss of buoyancy and settling to deeper water layers and the seabed (20-24), rapid stranding on beaches (25, 26), degradation through physical fragmentation (18), microbial alteration (27) and/or photochemical dissolution (28). None of these studies has nevertheless questioned the robustness of so short residence times of MPs at the ocean surface as those derived from the comparison between river fluxes and floating ocean stocks.

Contrary to measurements at sea, MP studies in rivers were scarce at the time of initial river flux estimates and disregarded differences in sampling techniques. However, recent studies on MPs in rivers allowed us to reassess river fluxes based on a more exhaustive data compilation than previously made (Fig. 1 and Table S1). We consequently identify three main methodological biases that magnified errors in the process leading from the quantification of plastic debris in individual rivers to the calculation of global river budgets. Correcting these errors leads to flux estimates that are orders of magnitude (OM hereafter) smaller than previous values. The quest for a major missing plastic sink in the ocean becomes then needless.



**Fig. 1. Global watershed delineation (0.5 x 0.5 degree grid cell resolution) and representation of the river catchments for which MP flux estimates can be established from the literature.** They represent 340 measurements at 127 sampling locations in 89 rivers and have been compiled from 42 published studies (Table S1). With respect to the world total, the 89 sampled rivers represent 14% of water discharge.

5

The first source of error refers to the choice of a representative mass for MPs in rivers. Since most field studies only determined MP number concentrations, the accurate estimate of MP mass transport depends on knowing how to convert numbers to mass. Based on our extended dataset (n = 129), a median mass of 0.23 mg item<sup>-1</sup> (Weiss conversion factor 1, Wf1) for non-fiber MPs is obtained (Supplementary Materials and Methods 1.3 and Figs. S1, S2), which is significantly lower than those used in previous estimates, i.e. 3.0 mg item<sup>-1</sup> (7) or 0.96 mg item<sup>-1</sup> (8), referred as Lf and Sf conversion factors hereafter (according to initial letters of the first authors in (7) and (8), respectively). Our estimate for MP fibers is 0.75 µg item<sup>-1</sup> (Weiss conversion factor 2, Wf2). Since bulk samples can be largely composed by fibers (up to >90%, Table S1), composite MP mass (estimated using a composite Weiss factor, Wf) often represents only a small fraction of Wf1, and Wf frequently is 1-2 OM lower than Lf or Sf.

10

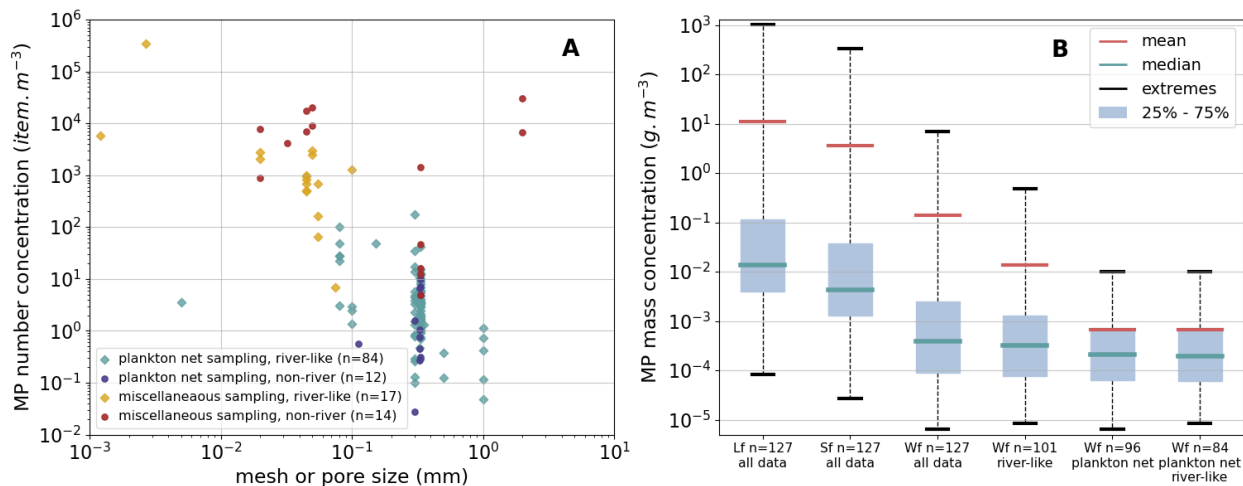
15

The second source of error results from the inappropriate mixing of data collected with different sampling techniques. MPs at sea are usually collected with plankton nets (29), which is also the most common sampling technique in rivers (30) (hereafter “plankton net sampling”). However, in a number of studies, MPs have been collected using other techniques (hereafter “miscellaneous

20

sampling”) including filtration through pore sizes smaller than the minimum mesh size of plankton nets (80  $\mu\text{m}$ ). Our data clearly show that filtration collects much larger numbers of MPs (Fig. 2A) and, particularly, small-sized MPs, which is consistent with previous studies that found distributions of MPs sampled with this method skewed toward the lower size range in rivers (31), atmospheric deposits (32) and seawater (33). Major biases arise when such small plastic fragments are included in mass conversions using factors obtained from plankton net sampling. A striking example is the Yangtze River for which miscellaneous sampling on filters (34) produced MP number concentrations 3 to 4 OM larger than sampling with plankton nets (35). The mass offset rises to almost 5 OM when the number concentrations based on filtration are converted to mass concentrations by using Lf [as applied in (7)] and the number concentrations based on plankton net sampling are converted by using Wf.

A box plot of MP mass concentrations highlights the combined effect of inappropriate conversion factors and misleading mixing of measurements from different sampling techniques (Fig. 2B). Applying uniformly the Lf factor to the reported number concentrations in our exhaustive data set yields a mass concentration range encompassing 7 OM. MP mass concentrations would be in some cases as low as those of natural trace metals during low flow conditions (36) while in other cases they would be as high as total suspended sediment concentrations during flood conditions (37). Such an extended mass range has never been reported for any kind of particulate matter in rivers before. Using however the Wf factor not only lowers the median mass concentration by more than a factor of 30, but also narrows its variability by one OM. Restricting the outcomes from field studies to those from plankton net sampling further lowers median concentrations and leads to the strongest reduction of variability. Values now only stretch over 3 OM, which is more in line with the maximum seasonal variability for MP concentrations reported in case studies (38-40).

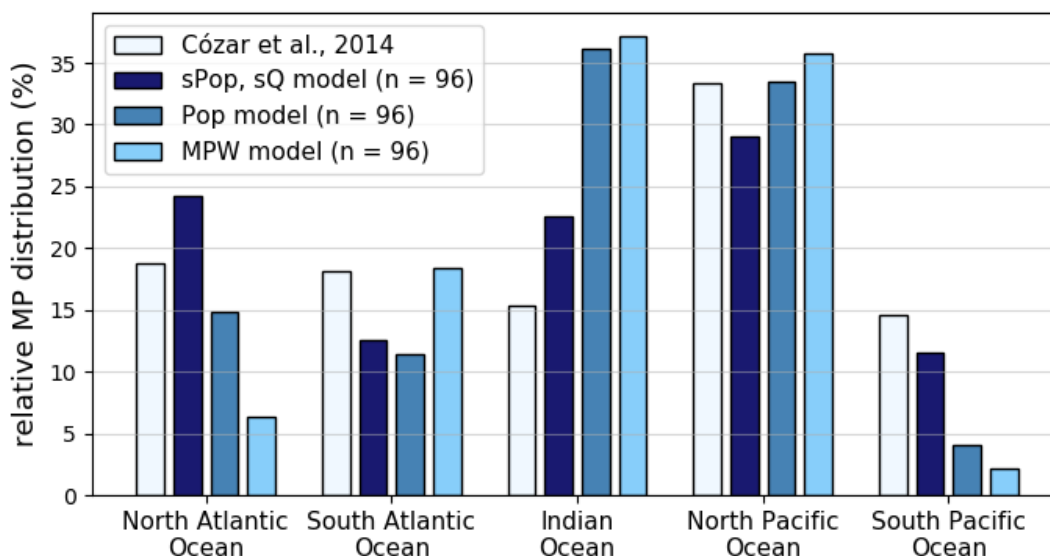


**Fig. 2. MP concentration variability in rivers.** (A) MP number concentrations (items  $\text{m}^{-3}$ ) in different data subsets combining sampling techniques and study locations. Retention limits correspond to mesh (plankton net sampling) or filter pore sizes (miscellaneous sampling) used to retain MP particles according to the compiled studies. “River-like” sampling considers data obtained in rivers *sensu stricto*, *i.e.* excluding connected surface waters such as artificial canals, channelized streams, tidal estuaries or bays (Supplementary Materials and Methods 1.2). (B) Box plot representation of MP mass concentrations as a function of different combinations of sampling techniques and number to mass conversion factors (Lf, Sf, Wf). The similarity of the two box plots at the right (all plankton net data and plankton net data restricted to rivers) indicates that the high variability observed in the other box plots is mostly related to miscellaneous sampling.

The third source of error concerns global MP flux estimates based on the relationship between MP river fluxes and MPW. Our calculations yield MP fluxes as high as millions of tons when using the above-mentioned Lf or Sf conversion factors (4,610 and 1,479 kt yr<sup>-1</sup> respectively) and correlating river mass fluxes with MPW stocks in drainage basins (Table 1). Nevertheless, the associated values of the coefficient of determination are quite low ( $r^2 = 0.53$ ), suggesting that the models are far from fully explaining the calculated river loads. Applying the composite factor Wf drops the global MP flux to 129 kt yr<sup>-1</sup>, while  $r^2$  increases to 0.59. Restricting the regression to the plankton net data subset further improves the correlation ( $r^2 = 0.66$ ) and reduces the global flux to 31 kt yr<sup>-1</sup>. Interestingly, with this latter subset of data, MPW is no longer the best predictor for MP river loads, since the associated coefficient of determination is greater ( $r^2 = 0.69$ ) if correlated to total basin population (Pop). Processing the large-scale extrapolations with the parametric equation based on regression with Pop instead of MPW still decreases the global MP flux by about one OM down to 3.5 kt yr<sup>-1</sup>. We also tested whether multi-regression models can improve the statistics (Supplementary Materials and Methods 2.2 and Table S3). For the plankton net dataset, we identify a significant relationship between specific MP fluxes, population density (sPop) and drainage intensity (sQ). This relationship increases again the coefficient of determination to  $r^2 = 0.80$  (Table 1), although global budgets only change moderately (6.1 kt yr<sup>-1</sup>). However, regression analyses alone do not allow rejection of MPW as the best predictor for riverine MP fluxes but the heterogeneity that MPW values implied in the spatial distribution of riverine MP discharges into the ocean does. Indeed, this heterogeneity does not match the observed accumulation patterns in the different ocean basins (Fig. 3). The discrepancy is particularly striking for the North Atlantic Ocean. Although this basin should have a MP concentration at least as large as the North Pacific Ocean (12, 33, 41), it always receives less than 6% of the global river inputs when extrapolations are calculated from MPW (Table 1). This is because MPW estimates for Europe and North America are quite low compared to Asia and Africa (Fig. S3). Extrapolations based on Pop and/or on sPop and sQ better fit with observed MP accumulation at the ocean surface (Fig. 3). Very recently, a modeling study (6) proposed a scenario of global river plastic fluxes based on a Human Development Index (HDI). Although the authors found clearly lower and spatially more equilibrated values compared to (7) and (8), discrepancies with the accumulation patterns at the ocean surfaces still exist. They are more important than in our approach, which is not giving more credit to HDI compared to Pop.

**Table 1. Regression models for observed MP river fluxes and potential controlling factors in the respective drainage basins for the data subsets shown in Fig. 2.** The resulting global MP fluxes are further detailed according to the main ocean basins to which they are discharged. The “Others” column includes discharges into the Mediterranean Sea, the Black Sea and the Arctic Ocean. MPW: Mismatched plastic waste; Pop: basin population; sPop: population density; sQ: drainage intensity.

Data subsets	Number of data	Regression parameters	Number to mass conversion factor	r <sup>2</sup>	Global MP flux (kt yr <sup>-1</sup> )	North Atlantic (%)	South Atlantic (%)	Indian Ocean (%)	North Pacific (%)	South Pacific (%)	Others (%)
All data	127	MPW	Lf	0.53	4,610	5.3	16.3	33.6	31.7	1.7	11.4
All data	127	MPW	Sf	0.53	1,479	5.3	16.3	33.6	31.7	1.7	11.4
All data	127	MPW	Wf	0.59	129	4.8	16.3	34.3	31.8	1.5	11.3
All data	127	Pop	Wf	0.49	15	12.6	10.1	33.4	30.2	3.3	10.4
River-like	101	MPW	Wf	0.57	56	6.1	16.3	32.3	31.7	2.1	11.5
River-like	101	Pop	Wf	0.45	7.2	15.4	10.5	29.0	29.3	4.9	10.9
Plankton net	96	MPW	Wf	0.66	31	5.7	16.3	32.9	31.7	1.9	11.5
Plankton net	96	Pop	Wf	0.69	3.5	13.3	10.2	32.3	30.0	3.7	10.5
Plankton net – River	84	MPW	Wf	0.66	29	6.0	16.3	32.4	31.7	2.1	11.5
Plankton net – River	84	Pop	Wf	0.69	3.6	13.8	10.3	31.5	29.8	4.0	10.6
Plankton net	96	sPop, sQ	Wf	0.80	6.1	23.1	12.0	21.6	27.7	11.0	4.6
Plankton net – River	84	sPop, sQ	Wf	0.80	6.6	23.3	12.0	21.4	27.6	11.0	4.7



**Fig. 3. Comparison between relative distribution of estimated MP stocks at the surface of the different ocean basins (12) and relative distribution of riverine MP fluxes according to the “plankton net sampling” model outputs (Table 1).** Both surface ocean stocks for the listed ocean basins and river fluxes close at 100%. Extrapolations of MP river fluxes based on Pop and/or sPop and sQ generally fit better with observed MPs accumulation at the ocean surface according to Cózar’s et al., observations (12) than extrapolation based on MPW. Nevertheless, discrepancies exist, especially for the Indian Ocean in the Southern Hemisphere. The oceanic gyre is here connected to the South Atlantic via the Agulhas Current leakage (45) and to the South Pacific via the Great Australian Bight and the Tasman Sea (46), suggesting that the stocks in these two ocean basins could be partly fed by MPs originating from the Indian Ocean.

Lowering the global river plastic flux down to a few thousand tons per year has of course major implications for our understanding of MPs cycling in the sea. Conservatively dividing the oceanic stock of 14.4 kt (12) by our best MP flux estimate of 6.1 kt yr<sup>-1</sup> yields residence times around 2.4 years for the floating stock at the ocean surface. Although associated with significant uncertainties (Supplementary Materials and Methods 3), such values support the idea that surface oceanic MP stocks result from rather slow accumulation processes and thus conflicts with the concept of ocean MPs being a highly reactive pool in terms of biological and chemical degradation and sequestration processes. Longer residence times are also in accordance with the observed long-distance transport of invasive species living on rafting plastics that may be part of the “plastisphere” (27, 42). Slow accumulation does not exclude selective sorting and sequestration of MPs on their way from land

to the oceanic gyres. The average mass of a floating oceanic MP is generally higher (between 0.97 and 7 mg (10, 12, 13, 18, 41)) than the value we found for non-fiber MPs in rivers and tends to increase in accumulation areas (Supplementary Materials and Methods 3, and Fig. S1). This difference could reflect selective sequestration of smaller MPs, for example through turbulent dispersion in the water column (33) or ingestion by a wide range of organisms (14, 17, 43) and subsequent transfer to seafloor sediments where they form the bulk of plastic litter (20, 23, 44).

Our finding that MP river fluxes are much lower than previous estimates should not be misunderstood as an attempt to minimize the severity of the plastic pollution problem. On the contrary, it implies that even if riverine plastic inputs were to cease immediately, the floating MP stocks and their deleterious effects on the marine environment would persist for many years. We neither claim that plastic cycling in the oceans is fully understood. There are many size classes, oceanic compartments and land-to-sea transfer processes on which further research is urgently needed to properly evaluate the exposure of marine biota to this novel type of pollution and design cleaning strategies. Reliable mass quantification of plastic stocks and fluxes both in the lower micrometer to nanometer scale and in the form of macroplastics are on top of the priority list for action. However, as we clearly demonstrate with our results, in all of these approaches, great effort should be spent on the development of standard sampling techniques and reliable extrapolation methods in order to avoid major errors in budget calculations that are uncritically accepted. We finally demonstrate that MP pollution in the world ocean is not restricted to MPW-designated hot spot countries in Asia or Africa. Collective initiatives aiming at reducing the ocean's MP load therefore should rely more on global land-based actions at much larger spatial scales than previously thought.

## REFERENCES

1. D. K. Barnes, F. Galgani, R. C., Thompson, M. Barlaz, Accumulation and fragmentation of plastic debris in global environments. *Philos. Trans. R. Soc. B Biol. Sci.* **364**, 1985-1998 (2009).
2. C. M. Rochman, T. Hoellein, The global odyssey of plastic pollution. *Science* **368**, 1184-1185 (2020).
3. Group of Experts on the Scientific Aspects of Marine Protection (GESAMP). Sources, fate and effects of microplastics in the marine environment: part two of a global assessment (eds P.J. Kershaw, C. M. Rochman). *Rep. Stud. GESAMP* **93**, 220 (2016).
4. J.R. Jambeck, R. Geyer, C. Wilcox, T. R. Siegler, M. Perryman, A. Andrady, R. Narayan, K. L. Law, Plastic waste inputs from land into the ocean. *Science* **347**, 768-771 (2015).
5. L. Lebreton, A. Andrady, Future scenarios of global plastic waste generation and disposal. *Palgrave Commun.* **5**, 6 (2019). Figshare <https://doi.org/10.6084/m9.figshare.5900335> (2018).
6. L. Mai, X.-F. Sun, L.-L. Xia, L.-J. Bao, L.-Y. Liu, E. Y. Zeng, Global Riverine Plastic Outflows. *Environ. Sci. Technol.* **54**, 10049-10056 (2020).
7. L. C. M. Lebreton, J. Van der Zwet, J. W. Damsteeg, B. Slat, A. Andrady, J. Reisser, River plastic emissions to the world's oceans. *Nat. Commun.* **8**, 15611 (2017).
8. C. Schmidt, T. Krauth, S. Wagner, Export of plastic debris by rivers into the sea. *Environ. Sci. Technol.* **51**, 12246-12253 (2017).
9. W. W. Y. Lau, Y. Shiran, R. M. Bailey, E. Cook, M. R. Stuchtey, J. Koskella, C. A. Velis, L. Godfrey, J. Boucher, M. B. Murphy, R. C. Thompson, E. Jankowska, A. C. Castillo, T. D. Pilditch, B. Dixon, L. Koerselman, E. Kosior, E. Favoino, J. Gutberlet, S. Baulch, M. E. Atreya, D. Fischer, K. K. He, M. M. Petit, U. R. Sumaila, E. Neil, M. V. Bernhofen, K. Lawrence, J. E. Palardy, Evaluating scenarios toward zero plastic pollution. *Science*, **369**, 1455-1461 (2020).
10. S. E. Morét-Ferguson, K. L. Law, G. Proskurowski, E. K. Murphy, E. E. Peacock, C. M. Reddy, The size, mass, and composition of plastic debris in the Western North Atlantic Ocean. *Mar. Pollut. Bull.* **60**, 1873-8 (2010).
11. E. van Sebille, C. Wilcox, L. Lebreton, N. Maximenko, B. D. Hardesty, V. J. A. Franeker, M. Eriksen, D. Siegel, F. Galgani, K. L. Law, A global inventory of small floating plastic debris. *Environ. Res. Lett.* **10**, 124006 (2015).



12. A. Cózar, F. Echevarría, J. I. González-Gordillo, X. Irigoien, B. Ubeda, S. Hernández-León, Plastic debris in the open ocean. *Proc. Natl. Acad. Sci.* **111**, 10239–10244 (2014).
13. M. Eriksen, L. C. M. Lebreton, H. S. Carson, M. Thiel, C. J. Moore, J. C. Borerro, F. Galgani, P. G. Ryan, J. Reisser, Plastic pollution in the world's oceans: more than 5 trillion plastic pieces weighing over 250,000 tons afloat at sea. *PLoS ONE* **9**, e111913 (2014).
14. K. L. Law, S. E. Morét-Ferguson, N. A. Maximenko, G. Proskurowski, E. E. Peacock, J. Hafner, C. M. Reddy, Plastic accumulation in the North Atlantic subtropical gyre. *Science* **329**, 1185–88 (2010).
15. K. L. Law, S. E. Morét-Ferguson, D. S. Goodwin, E. R. Zettler, E. DeForce, T. Kukulka, G. Proskurowski, Distribution of surface plastic debris in the eastern Pacific Ocean from an 11-year data set. *Environ. Sci. Technol.* **48**, 4732–38 (2014).
16. C. D. Keeling, S. C. Piper, M. Heimann, A three-dimensional model of atmospheric CO<sub>2</sub> transport based on observed winds: 4. Mean annual gradients and interannual variations. In D. H. Peterson (Ed.), *Aspects of Climate Variability in the Pacific and the Western Americas*. Washington: AGU. *Geophys. Monogr.* **55**, 305–63 (1989).
17. R. C. Thompson, Y. Olsen, R. P. Mitchell, A. Davis, S. J. Rowland, A. W. G. John, D. McGonigle, A. E. Russell, Lost at Sea: Where Is All the Plastic? *Science* **304**, 838 (2004).
18. A. ter Halle, L. Ladirat, X. Gendre, D. Goudouneche, C. Pusineri, C. Routaboul, C. Tenailleau, B. Duployer, E. Perez, Understanding the fragmentation pattern of marine plastic debris. *Environ. Sci. Technol.* **50**, 5668–5675 (2016).
19. P. Davison, R. G. Asch, Plastic ingestion by mesopelagic fishes in the North Pacific Subtropical Gyre. *Mar. Ecol. Prog. Ser.* **432**, 173–80 (2011).
20. L. C. Woodall, A. Sanchez-Vidal, M. Canals, G. L. J. Paterson, R. Coppock, V. Sleight, A. Calafat, A. D. Rogers, B. E. Narayanaswamy, R. Thompson, The deep sea is a major sink for microplastic debris. *R. Soc. Open Sci.* **1**, 140317 (2014).
21. W. P. de Haan, A. Sanchez-Vidal, M. Canals, on behalf of the NUREIEV1 Shipboard Scientific Party, Floating microplastics and aggregate formation in the Western Mediterranean Sea. *Mar. Pollut. Bull.* **140**, 523–535 (2019).
22. K. Pabortsava, R. S. Lampitt, High concentrations of plastic hidden beneath the surface of the Atlantic Ocean. *Nat. Commun.* **11**:4073 (2020).
23. I. A. Kane, M. A. Clare, E. Miramontes, R. Wogelius, J. J. Rothwell, P. Garreau, F. Pohl, Seafloor microplastic hotspots controlled by deep-sea circulation. *Science* **10**, 1126 (2020).
24. J. Boucher, G. Billard, *The Mediterranean: Mare plasticum*. Gland, Switzerland: IUCN. x+62 pp
25. S. Liubartseva, G. Coppini, R. Lecci, E. Clementi, Tracking plastics in the Mediterranean: 2D Lagrangian model. *Mar. Pollut. Bull.* **129**, 151–162 (2018).
26. Y. Ourmières, J. Mansui, A. Molcard, F. Galgani, I. Poitou, The boundary current role on the transport and stranding of floating marine litter: The French Riviera case. *Cont. Shelf Res.* **155**, 11–20 (2018).
27. J. Jacquin, J. Cheng, C. Odobel, C. Pandin, P. Conan, M. Pujon-Pay, V. Barbe, A. L. Meistertzheim, J. F. Ghiglione, Microbial ecotoxicology of marine plastic debris: a review on colonization and biodegradation by the “plastisphere”. *Front. Microbiol.* **10**, 865 (2019).
28. L. Zhu, S. Zhao, T. B. Bittar, A. Stubbins, D. Li, Photochemical dissolution of buoyant microplastics to dissolved organic carbon: Rates and microbial impacts. *J. Haz. Mat.* **383**, 121065 (2020).
29. V. Hidalgo-Ruz, L. Gutow, R. C. Thompson, M. Thiel, Microplastics in the Marine Environment: a review of the methods used for identification and quantification. *Environ. Sci. Technol.* **46**, 3060–3075 (2012).
30. R. Dris, H. Imhof, W. Sanchez, J. Gasperi, F. Galgani, B. Tassin, C. Laforsch, Beyond the ocean: contamination of freshwater ecosystems with (micro-) plastic particles. *Environ. Chem.* **12**, 539–550 (2015).
31. R. Dris, J. Gasperi, V. Rocher, M. Saad, N. Renault, B. Tassin, Microplastic contamination in an urban area: a case study in Greater Paris. *Environ. Chem.* **12**, 592–599 (2015).
32. S. Allen, D. Allen, V. R. Phoenix, G. L. Roux, P. D. Jiménez, A. Simonneau, S. Binet, D. Galop, Atmospheric transport and deposition of microplastics in a remote mountain catchment. *Nat. Geosci.* **12**, 339 - 344 (2019).
33. M. Poulain, M. J. Mercier, L. Brach, M. Martignac, C. Routaboul, E. Perez, M. C. Desjean, A. ter Halle, Small Microplastics as a Main Contributor to Plastic Mass Balance in the North Atlantic Subtropical Gyre. *Environ. Sci. Technol.* **53**, 1157–1164 (2019).
34. S. Zhao, L. Zhu, T. Wang, D. Li, Suspended microplastics in the surface water of the Yangtze estuary system, China: first observations on occurrence, distribution. *Mar. Pollut. Bull.* **86**, 562–568 (2014).
35. X. Xiong, C. Wu, J. J. Elser, Z. Mei, Y. Hao, Occurrence and fate of microplastic debris in middle and lower reaches of the Yangtze River – From inland to the sea. *Sci. Total Environ.* **659**, 66–73 (2018).
36. J. M. Martin, M. Meybeck, Elemental Mass-Balance of Material Carried by Major World Rivers. *Mar. Chem.* **7**, 173–206 (1979).

37. M. Meybeck, L. Laroche, H. H. Dürr, J. P. M. Syvitski, Global variability of daily total suspended solids and their fluxes in rivers. *Glob. Planet. Change* **39**, 65–93 (2003).
38. M. Constant, W. Ludwig, P. Kerhervé, J. Sola, B. Charrière, A. Sanchez-Vidal, M. Canals, S. Heussner, Microplastic fluxes in a large and a small Mediterranean river catchments: The Têt and the Rhône, Northwestern Mediterranean Sea. *Sci. Total Environ.* **716**, 136984 (2020).
39. F. Faure, C. Demars, O. Wieser, M. Kunz, L. F. de Alencastro, Plastic pollution in Swiss surface waters: nature and concentrations, interaction with pollutants. *Environ. Chem.* **12**, 582–591 (2015).
40. A. Lechner, H. Keckeis, F. Lamesberger-Loisl, B. Zens, R. Krusch, M. Tritthart, M. Glas, E. Schludermann, The Danube so colourful: a potpourri of plastic litter outnumbers fish larvae in Europe's second largest river. *Environ. Pollut.* **188**, 177-181 (2014).
41. L. Lebreton, B. Slat, F. Ferrari, B. Sainte-Rose, J. Aitken, R. Marthouse, S. Hajbane, S. Cunsolo, A. Schwarz, A. Levivier, K. Noble, P. Debeljak, H. Maral, R. Schoeneich-Argent, R. Brambini, J. Reisser, Evidence that the Great Pacific Garbage Patch is rapidly accumulating plastic. *Scientific Report* **8**, 4666 (2018).
42. E. R. Zettler, T. J. Mincer, L. A. Amaral-Zettler, Life in the “Plastisphere”: microbial communities on plastic marine debris. *Environ. Sci. Technol.* **47**, 7137–46 (2013).
43. S. Deudero, C. Alomar, Mediterranean marine biodiversity under threat: reviewing influence of marine litter on species. *Mar. Pollut. Bull.* **98**, 58-68 (2018).
44. W. Courtene-Jones, B. Quinn, S. F. Gary, A. O. M. Mogg, Microplastic pollution identified in deep-sea water and ingested by benthic invertebrates in the Rockall Trough, North Atlantic Ocean. *Environ. Pollut.* **231**, 271-280 (2017).
45. W. P. M. De Ruijter, A. Biastoch, S. S. Drijfhout, J. R. E. Lutjeharms, R. P. Matano, T. Pichevin, P. J. Leeuwen, W. Weijer, Indian–Atlantic interocean exchange: Dynamics, estimation and impact. *J. Geophys. Res.* **104**, 885–20,910 (1999).
46. C. Maes, N. Grima, B. Blanke, E. Martinez, T. Paviet-Salomon, T. Huck,. A surface “superconvergence” pathway connecting the South Indian Ocean to the subtropical South Pacific gyre. *Geophys. Res. Lett.* **45**, 1915–1922 (2018).
47. C. J. Moore, G. L. Lattin, A. F. Zellers, Density of Plastic Particles found in zooplankton trawls from Coastal Waters of California to the North Pacific Central Gyre. Proceedings of the Plastic Debris Rivers to Sea Conference, Algalita Marine Research Foundation, Long Beach, CA USA (2005).
48. A. L. Andrady, *Plastics and the environment*. In: Anthony L. Andrady (Ed.), Publisher: John Wiley & Sons (2004).
49. C. J. Moore, G. L. Lattin, A. F. Zellers, Quantity and type of plastic debris flowing from two urban rivers to coastal waters and beaches of Southern California. *J. Integr. Coast. Zone Manag.* **11**, 65–73 (2011).
50. S. S. Sadri, R. C. Thompson, On the quantity and composition of floating plastic debris entering and leaving the Tamar Estuary, Southwest England. *Mar. Pollut. Bull.* **81**, 55-60 (2014).
51. J. Gasperi, R. Dris, T. Bonin, V. Rocher, B. Tassin, Assessment of floating plastic debris in surface water along the Seine River. *Environ. Pollut.* **195**, 163-166 (2014).
52. M. Constant, P. Kerhervé, M. Mino-Vercellio-Verollet, M. Dumontier, A. Sánchez-Vidal, M. Canals, S. Heussner, Beached microplastics in the northwestern Mediterranean Sea. *Mar. Pollut. Bull.* **142**, 263–273 (2019).
53. L. T. Yonkos, E. A. Friedel, A. C. Perez-Reyes, S. Ghosal, C. D. Arthur, Microplastics in four estuarine rivers in the Chesapeake Bay, USA. *Environ. Sci. Technol.* **48**, 14195-14202 (2014).
54. M. van der Wal, M. van der Meulen, G. Tweehuijsen, M. Peterlin, A. Palatinus, M. K. Virsek, L. Coscia, A. Krzan, Identification and Assessment of Riverine Input of (Marine) Litter. Final Report for the European Commission DG Environment under Framework Contract No ENV.D.2/FRA/2012/0025 (2015).
55. M. O. Rodrigues, N. Abrantes, F. J. M. Goncalves, H. Nogueira, J. C. Marques, A. M. M. Goncalves, Spatial and temporal distribution of microplastics in water and sediments of a freshwater system (Antua River, Portugal). *Sci. Total Environ.* **633**, 1549-1559 (2018).
56. L. Lahens, E. Strady, T. C. Kieu Le, R. Dris, B. Kada, E. Rinnert, J. Gasperi, B. Tassin, Macroplastic and microplastic contamination assessment of a tropical river (Saigon River, Vietnam) transversed by a developing megacity. *Environ. Pollut.* **236**, 661-671 (2018).
57. T. Kataoka, Y. Nihei, K. Kudou, H. Hinata, Assessment of the sources and inflow processes of microplastics in the river environments of Japan. *Environ. Pollut.* **244**, 958–965 (2018).
58. Center for International Earth Science Information Network - CIESIN - Columbia University. 2018. 30 arc-second resolution Gridded Population of the World for the year 2015, Version 4 (GPWv4): Population Density, Revision 11. Palisades, NY: NASA Socioeconomic Data and Applications Center (SEDAC). [https://doi.org/10.7927/H49C6VHW\\_\(2018\)](https://doi.org/10.7927/H49C6VHW_(2018)).

59. W. Ludwig, P. Amiotte-Suchet, G. Munhoven, J. L. Probst, Atmospheric CO<sub>2</sub> consumption by continental erosion: Present-day controls and implications for the last glacial maximum. *Glob. Planet. Change* **16**, 95-108 (1998a).
60. W. Ludwig, J. L. Probst, River sediment discharge to the oceans: Present-day controls and global budgets. *American J. Sci.* **296**, 265-295 (1998b).
61. P. Döll, B. Lehner, Validation of a new global 30-min drainage direction map. *J. Hydrol.* **258**, 214–231 (2002).
62. R. Dris, J. Gasperi, M. Saad, C. Mirande, B. Tassin, Synthetic fibers in atmospheric fallout : A source of microplastics in the environment? *Mar. Pollut. Bull.* **104**, 290-293 (2016).
63. L. Cai, J. Wang, J. Peng, Z. Tan, Z. Zhan, X. Tan, Q. Chen, Characteristic of microplastics in the atmospheric fallout from Dongguan city, China: Preliminary research and first evidence. *Environ. Sci. Pollut. Res.* **24**, 24928-24935 (2017).
64. M. Trainic, J. M. Flores, I. Pinkas, M. L. Pedrotti, F. Lombard, G. Bourdin, G. Gorsky, E. Boss, Y. Rudich, A. Vardi, I. Koren, Airborne microplastic particles detected in the remote marine atmosphere. *Commun. Earth Environ.* **1**, 64 (2020).
65. J. van Wijnen, A. M. Ragas, C. Kroeze, Modelling global river export of microplastics to the marine environment: Sources and future trends. *Sci Total Environ.* **673**, 392-401 (2019).
66. M. Kooi, J. Reisser, B. Slat, F. F. Ferrari, M. S. Schmid, S. Cunsolo, R. Brambini, K. Noble, L.-A. Sirks, T. E. W. Linders, R. I. Schoeneich-Argent, A. A. Koelmans, The effect of particle properties on the depth profile of buoyant plastics in the ocean. *Sci. Rep.* **6**, 1-10 (2016).
67. D. Kushary, Bootstrap methods and their application. *Technometrics* **42**, 216-217 (2000).
68. A. K. Baldwin, S. R. Corsi, S. A. Mason, Plastic Debris in 29 Great Lakes Tributaries: Relations to Watershed Attributes and Hydrology. *Environ. Sci. Technol.* **50**, 10377–10385 (2016).
69. T. Mani, A. Hauk, U. Walter, P. Burkhardt-Holm, Micro-plastics profile along the Rhine River. *Sci. Rep.* **5**, 17988 (2016).
70. W. Wang, A. W. Ndungu, Z. Li, J. Wang, Microplastics pollution in inland freshwaters of China: A case study in urban surface waters of Wuhan, China. *Sci. Total Environ.* **575**, 1369–1374 (2017).
71. A. R. McCormick, T. J. Hoellein, S. A. Mason, J. Schlupe, J. J. Kelly, Microplastic is an Abundant and Distinct Microbial Habitat in an Urban River. *Environ. Sci. Technol.* **48**, 11863–11871 (2014).
72. A. R. McCormick, T. J. Hoellein, M. G. London, J. Hittie, J. W. Scott, J. J. Kelly, Microplastic in surface waters of urban rivers: concentration, sources, and associated bacterial assemblages. *Ecosphere* **7**, e01556 (2016).
73. S. Rech, V. Macaya-Caquilpan, J. F. Pantoja, M. M. Rivadeneira, C. K. Campodonico, M. Thiel, Sampling of riverine litter with citizen scientists - findings and recommendations. *Environ. Monit. Assess.* **187**, (2015).
74. A. Vianello, F. Acri, F. B. Aubry, A. Boldrin, E. Camatti, L. Da Ros, T. Marceta, V. Moschino, Occurrence and distribution of floating microplastics in the North Adriatic Sea: preliminary results. Proceedings of the MICRO2015 Seminar on microplastics (Piran, 2015).
75. S. Estahbanati, N. L. Fahrenfeld, Influence of wastewater treatment plant discharges on microplastic concentration in surface water. *Chemosphere* **162**, 277-284 (2016).
76. A. R. A. Lima, M. F. Costa, M. Barletta, Distribution patterns of microplastics within the plankton of tropical estuary. *Environ. Res.* **132**, 146-155 (2014).
77. P. Hohenblum, B. Liebmam, M. Liedermann, Plastic and microplastic in the environment. Umweltbundesamt REP-0551, Vienna (2015).
78. J. C. Vermaire, C. Pomeroy, S. M. Herczegh, O. Haggart, M. Murphy, Microplastic abundance and distribution in the open water and sediment of the Ottawa River, Canada, and its tributaries. *FACETS* **2**, 301–314 (2017).
79. R. Z. Miller, A. J. R. Watts, B. O. Winslow, T. S. Galloway, A. P. W. Barrows, Mountains to the sea: River study of plastic and non-plastic microfiber pollution in the northeast USA. *Mar. Pollut. Bull.* (2017).
80. S. H. Campbell, P. R. Williamson, B. D. Hall, Microplastics in the gastrointestinal tracts of fish and the water from an urban prairie creek. *FACETS* **2**, 395–409 (2017).
81. L. K. Schmidt, M. Bochow, H. K. Imhof, S. E. Oswald, Multi-temporal surveys for microplastic particles enabled by a novel and fast application of SWIR imaging spectroscopy - Study of an urban watercourse traversing the city of Berlin, Germany. *Environ. Pollut.* **239**, 579-589 (2018).
82. L. Lin, L. Z. Zuo, J. P. Peng, L. Q. Cai, L. Fok, Y. Yan, H. X. Li, X. R. Xu, Occurrence and distribution of microplastics in an urban river: A case study in the Pearl River along Guangzhou City, China. *Sci. Total Environ.* **644**, 375-381 (2018).
83. K. J. Kapp, E. Yeatman, Microplastic hotspots in the Snake and Lower Columbia rivers: A journey from the greater Yellowstone ecosystem to the Pacific Ocean. *Environ. Pollut.* **241**, 1082–1090 (2018).

84. R. Dris, J. Gasperi, V. Rocher, B. Tassin, Synthetic and non-synthetic anthropogenic fibers in a river under the impact of Paris Megacity: sampling methodological aspects and flux estimations. *Sci. Total Environ.* **618**, 157-164 (2018).
85. A. P. W. Barrows, K. S. Christiansen, E. T. Bode, T. J. Hoellein, A watershed- scale, citizen science approach to quantifying microplastic concentration in a mixed land-use river. *Water Res.* **147**, 382–392 (2018).
86. M. Yan, H. Nie, K. Xu, Y. He, Y. Hu, Y. Huang, J. Wang, Microplastic abundance, distribution and composition in the Pearl River along Guangzhou city and Pearl River estuary, China. *Chemosphere* **217**, 879-886 (2019).
87. L. Watkins, S. McGrattan, P. J. Sullivan, M. T. Walter, The effect of dams on river transport of microplastic pollution. *Sci. Total Environ.* **664**, 834-840 (2019).
88. X. Tan, X. Yu, L. Cai, J. Wang, J. Peng, Microplastic and associated PAHs in surface water from the Feilaixia Reservoir in the Beiji River, China. *Chemosphere* **221**, 834–840 (2019).
89. L. Simon-Sánchez, M. Gerlaud, J. Garcia-Orellana, P. Ziveri, River Deltas as hotspots of microplastic accumulation: The case study of the Ebro River (NW Mediterranean). *Sci. Total Environ.* **687**, 1186-1196 (2019).
90. A. D. Gray, H. Wertz, R. R. Leabs, J. E. Weinstein, Microplastic in two South Carolina Estuaries: Occurrence, distribution, and composition. *Mar. Pollut. Bull.* **128**, 223-233 (2018).
91. Z. T. Anderson, A. B. Cundy, I. W. Croudace, P. E. Warwick, O. Celis-Hernandez, J. L. Stead, A rapid method for assessing the accumulation of microplastics in the sea surface microlayer (SML) of estuarine systems. *Scientific Reports* **8**, 9428 (2018).
92. W. Luo, L. Su, N. J. Craig, F. Du, C. Wu, H. Shi, Comparison of microplastic pollution in different water bodies from urban creeks to coastal waters. *Environ. Pollut.* **246**, 174-182 (2018).
93. C. Jiang, L. Yin, Z. Li, X. Wen, X. Luo, S. Hu, H. Yang, Y. Long, B. Deng, L. Huang, Y. Liu, Microplastic pollution in the rivers of the Tibet Plateau. *Environ. Pollut.* **249**, 91-98 (2019).
94. L. Ding, R. fan Mao, X. Guo, X. Yang, Q. Zhang, C. Yang, Microplastics in surface waters and sediments of the Wei River, in the northwest of China. *Sci. Total Environ.* **667**, 427-434 (2019).
95. C. C. Deocarís, J. O. Allosada, L. T. Ardiente, L. G. G. Bitang, C. L. Dulohan, J. K. I. Lapuz, L. M. Padilla, V. P. Ramos, J. B. P. Padolina, Occurrence of microplastic fragments in the Pasig River. *H2O Open J.* **2**, 92-100 (2019).
96. F. C. Alam, E. Sembiring, B. S. Muntalif, V. Suendo, Microplastic distribution in surface water and sediment river around slum and industrial area (case study: Ciwalengke River, Majalaya district, Indonesia). *Chemosphere* **224**, 637-645 (2019).

## ACKNOWLEDGMENTS

We thank the anonymous peer reviewers for their suggestions and comments, which improved this manuscript. We are indebted to P. Döll who kindly supplied the WATERGAP river routing file in a 0.5-degree grid cell resolution and to S. Pinel for performing the calculations with the bootstrap statistical method. **Funding:** This work was supported by doctorate school ED305 of the University of Perpignan Via Domitia through attribution of PhD grants to L.W. and M.Co. M.Ca. acknowledges support from *Generalitat de Catalunya* 2017 SGR 315 grant to GRC Marine Geosciences, *Red Española sobre BASuras MARinas* (BAMAR, ref. CGL2016-81854-REDT) and *Institut d'Estudis Catalans* (IEC) I-MarCat grant (ref. PR2015-S06-CANALS). **Author contributions:** L.W. collected and assembled the MP data in rivers from the literature, performed the statistics, implemented the river GIS, and extrapolated and analyzed the global MP fluxes. W.L. steered the study design, conducted the discussion and contributed to data analysis. W.L., L.W. and S.H. wrote the manuscript with significant contributions from M.Ca., J-F.G. and C.E in their respective domains of expertise. M.Co. contributed to data mining from literature. M.Co., P.K. and M.Ca. supplied individual data on riverine MPs from previous studies. All authors discussed the results and commented on the manuscript. **Competing interests:** The authors declare no competing interests. **Data and materials availability:** The data frames used for the analyses are provided as Table S1 and in the Supplementary Materials and Methods and also are freely available online at the links specified in the references. The code for statistical analyses is based

on the *OLS statsmodels* and *scikit-learn* Python libraries (online documentation hosted at statsmodels.org and scikit-learn.org).

## **SUPPLEMENTARY MATERIALS**

- 5 Supplementary Materials and Methods
- References (47-96)
- Figs. S1-S3
- Tables S1-S3

# Supplementary Materials for

## **The missing ocean plastic sink: gone with the rivers**

Lisa Weiss, Wolfgang Ludwig, Serge Heussner, Miquel Canals, Jean-François Ghiglione,  
Claude Estournel, Mel Constant, Philippe Kerhervé

Correspondence to: [lisa.weiss@univ-perp.fr](mailto:lisa.weiss@univ-perp.fr)

### **This PDF file includes:**

Supplementary Materials and Methods

Supplementary References (47-96)

Figs. S1-S3

Tables S1-S3

## SUPPLEMENTARY MATERIALS AND METHODS

### 1. Data collection on MPs in rivers

#### 1.1. Data mining

Our dataset on microplastics (MPs) in rivers has been compiled based on an extensive bibliographic search and a careful analysis of the collected data (Table S1). All retrieved values have been thoroughly verified in the respective original source publications. The resulting database contains 127 average MP concentration values corresponding to 127 sampling stations from 42 published studies on 89 rivers and connected surface waters all over the world (Fig. 1). Most of them are located in Europe, North America and South-East Asia. Our study goes far beyond previously published compilations (7, 8) in terms of number of measurements ( $n = 340$ ) and diversity of studied environments and protocols. Our 127 average values have been calculated from these 340 referenced measurements. Repeated seasonal measurements for a given river station have been averaged to one single data point. For conversion to instantaneous fluxes we have used, whenever possible, the reported literature values for water discharges and basin areas. In many case studies, however, the associated water discharge was not cited, and we have used instead the long-term estimate extracted from our drainage intensity data layer (see below). Research on plastic pollution in rivers is recent. The first papers on this topic were published in 2004 and 2005 (17, 47, 48). It took another decade to develop the first field protocols for quantifying MPs in rivers [Danube (40), Los Angeles rivers (49), Tamar (50), Seine (51)]. Up to now, there is no standardized protocol and our database encompasses a great diversity of sampling and analytical procedures. Mesh or pore sizes for recovering MPs in freshwater samples are highly variable (3  $\mu\text{m}$  to 2 mm).

#### 1.2. Data subsets

In order to test whether the diversity of the sampled environments and the applied sampling methods could influence the calculation of MP concentrations and global budgets, collected data have been categorized according to two binary classifications. On the one hand, river studies have been classified according to sampling locations (“river-like” column, Table S1). Sampling in rivers *sensu stricto* (flag = 1,  $n = 101$ ) have been differentiated from other locations (flag = 0,  $n = 26$ ), such as artificial canals, channelized streams, tidal estuaries or connected bays. On the other hand, data have been classified according to sampling techniques (“plankton net” column, Table S1). Here, we distinguished sampling based on trawling with plankton nets (flag = 1,  $n = 96$ ), which is the most common technique used for MP sampling in freshwaters and open ocean environments (29, 30). The other methods (flag = 0,  $n = 31$ ) comprise continuous pumping through filters, bulk sampling with bottles and successive filtration, and/or micro-layer sampling with sieves. These methods normally use much smaller mesh or pore sizes to retain MP particles compared to plankton nets (i.e. 80–1000  $\mu\text{m}$ , with most of them corresponding to 300 or 333  $\mu\text{m}$ , e.g. manta-trawl).

#### 1.3. Conversion to mass concentrations

MP quantities in the literature are generally expressed as concentrations by numbers (i.e. number of items per unit volume), and mass concentrations are rarely provided. Only 8 of the 42 articles

in our data compilation provide measured mass concentrations. Thus, standard conversion factors had to be determined to calculate MP mass concentrations for the remaining 34 studies. To do so we have used the 8 studies that simultaneously measured both number and mass concentrations, separating the fraction of synthetic fibers from non-fiber MPs (such as fragments, beads, films, foams). Indeed, according to Constant et al. (38, 52), synthetic fibers account for more than 90% of the number concentrations, but only for 4 to 7% of the mass concentrations. Ideally, conversions should be performed specifically for the different size classes, shapes and polymer types, but the information that would be needed to do so is lacking.

Based on 129 samples for non-fiber MPs [from refs. (38, 39, 49, 53-57)], and 96 samples for fiber MPs [from refs. (38, 54-56)], we calculated an average particle mass per sample dividing the mass concentration by the number concentration. Then, we analyzed the distribution of average particle masses for the two shape categories, fiber and non-fiber MPs (Fig. S1A, B). Both histograms followed a normal distribution, although the fiber histogram was more irregular, which might be related to the smaller number of related studies. A probability density function was fitted to the data and showed that the maximum probabilities were approximately met at median particle masses of  $Wf1 = 0.23 \text{ mg item}^{-1}$  for non-fiber MPs and  $Wf2 = 0.75 \text{ } \mu\text{g item}^{-1}$  for fiber MPs. We therefore retained these values for number to mass conversions. The composite MP mass ( $Wf$ ) was consequently determined according to the following equation:

$$Wf = ( Wf1 * Cnf + Wf2 * Cf ) / ( Cnf + Cf )$$

with  $Cnf$  and  $Cf$  being the number concentrations ( $\text{item m}^{-3}$ ) of non-fiber and fiber MPs, respectively. As expected, conversions with  $Wf$  fit much better observations (Fig. S2A-D) compared to the previously proposed factors of  $Lf = 3.00 \text{ mg item}^{-1}$  (5) and  $Sf = 0.96 \text{ mg item}^{-1}$  (6).

## 2. Regression analysis

### 2.1. Potential predictor factors

For extrapolating riverine MP fluxes to regional and global scales, we performed regression analyses to find the best predictors among a series of georeferenced and globally available gridded parameters such as MPW (5), Pop (58), sPop or/and sQ (59). Previous modeling studies correlated the observed river MP fluxes (in tons per year) with MPW (4, 7, 8) alone, or MPW together with drainage intensity sQ (7) in their respective drainage basins. For MPW we used the newer data set of Lebreton and Andrady (5), although for ease of comparison also the older data set of Jambeck et al. (4) was considered. This data set was originally used in the modeling studies cited above. For sQ we took the data set of Ludwig et al. (59) and further integrated data sets on population numbers in 2015 (Pop) (58) and riverine particulate matter fluxes in general (60). Their spatial variability at global scales is summarized in Table S2 according to the main ocean basins to which they are connected.

Quantification of MPW stocks on land is strongly linked to total population numbers, but also depends on socio-economic behavior of the populations. MPW is hence more variable than Pop (Fig. S3). In the study of Lebreton and Andrady (5), MPW was estimated at much finer spatial



scales than in the original study of Jambeck et al. (4), rendering this dataset more suitable for our purpose. Both data sets are nevertheless highly correlated and have in common that they predict relatively low MPW stocks in the drainage basins connected to the North Atlantic (Table S2). Waste management is considered sophisticated in North America and Europe, reducing waste leakage rates to the environment. The two studies differ, however, on their evaluations of MPW stocks in drainage basins of the North Pacific, which is proportionally a larger hot spot for Jambeck et al. (4) than for Lebreton and Andrady (5). It should also be noticed that the latter study predicts a global MPW stock that is only about two thirds of the predicted MPW stock in the former study.

We extracted the average values for each potential predictor factor upstream of river sampling stations for all basins. Regressions were calibrated with our dataset of observed MP fluxes. The resulting equations were applied to the predictor parameters of each river basin worldwide in order to obtain the corresponding MP fluxes. For these extrapolations of average river MP fluxes at regional and global scales we used a simplified river routing scheme (61) allowing both identifying the main world river basins in a 0.5x0.5 degree grid cell resolution (30 arc minutes), and conservatively tracking riverine matter fluxes from headwaters down to the receiving ocean basins. This approach has the advantage that many global environmental data sets exist at this resolution, which consequently eases combinations for extrapolation purposes, at the cost of spatial resolution. For characterization of the 127 studied catchments, a second drainage basin data layer was created for the drainage basins and sub-basins holding stations that had been sampled for MPs (Fig. 1). Both were delineated by including all basin grids upstream of sampling points.

## 2.2. Multi-parametric regressions

A single controlling parameter may not be sufficient to explain the observed variability of MP fluxes in rivers, and multi-regression models can help identifying additional controlling factors. We therefore tested this type of regression based on specific, *i.e.* normalized by basin area, values for MP fluxes (sMP) and drainage basin characteristics (sMPW, sPop, sQ). Regression analyses with absolute values are strongly dominated by observations in the largest river basins, which can consequently be a source of statistical bias. When restricting the data to our plankton net data subsets, we find that the following model best fits our observations:

$$sMP = b * sPop^{a1} * sQ^{a2}$$

The regression coefficients a1, a2 and b are listed in Table S3. Correlation is rather weak for both data subsets ( $r^2 = 0.35$  and  $r^2 = 0.38$ ) but is significant for all parameters. Applying this model and converting the values into absolute MP fluxes increases the correlations in Table 1 to  $r^2 = 0.80$ . In addition, in our specific multi-regression models, sPop is a more powerful predictor for sMP than sMPW.

## 3. Constraints and uncertainty assessment

### 3.1 Residence time

Our approach of estimating plastic residence time in the surface ocean only considers the MP size fraction of plastic debris. Both MP sampling in the oceans and MP sampling in freshwaters rely

on common techniques (plankton net trawling) and sufficiently large data sets could be assembled to produce reliable quantitative estimates at the global scale. The surface ocean stocks were determined from hundreds to thousands of MP plankton net samples (11-13) and extrapolated together with oceanic circulation models to distinguish between accumulation (i.e. gyres) and non-accumulation zones. Also, most plastic flux estimates in rivers rely on measurements of MP concentrations. They are clearly more frequent than measurements of macroplastic (>5 mm) concentrations (7, 8) and often serve as extrapolation basis for the calculation of total plastic fluxes assuming constant MPs to macroplastics ratios (6, 7). Often macroplastic fluxes in rivers were not determined independently.

However, accumulation of plastic at the ocean surfaces also involves larger plastics and other sources than rivers. This introduces uncertainty into our results, which has to be addressed.

### 3.1.1 Size limits

Variable or unclear definition of the size limits between MPs, macroplastics and total plastics both in the surface ocean stocks and in the river fluxes is definitively a source of uncertainty. On the one hand, plankton net sampling of MPs in the oceans not always excluded plastic debris larger than 5 mm (as in rivers), but used instead an upper size limit of 100 mm (12) or 200 mm (13). Only one study (13) quantified this effect in terms of mass stocks and concluded that plastic debris in the 5-200 mm range contributed with about 43% to the total (<200 mm) stock collected by plankton net sampling. Macroplastics >200 mm such as plastic bottles are not included in the surface stock estimates in our study because they were determined by field observation surveys and not by plankton net sampling (13).

On the other hand, previous mass flux estimates from rivers were often only reported in the form of total plastics without further discrimination between MPs and macroplastics (e.g. 6, 7). Based on compilations of field studies that reported independently riverine MP and macroplastic mass concentrations for individual rivers, it can be roughly estimated that MPs represent on average about half of the total plastic mass concentrations (41% according to (7) with n=6 and 48% according to (8) with n=33) in rivers. Since a few larger particles can contribute a large fraction of the total plastic mass concentration (8), estimating this contribution is nevertheless problematic. Assuming that up to half of the MP mass stock at the ocean surface could be composed by particles in the 5–200 mm sizerange, it may therefore be concluded that size effects might represent an error factor of  $\leq 2$  in the calculation of average residence times based on MP river loads and surface ocean stocks, and of  $\leq 4$  based on total river loads (which may contain, although associated with large uncertainty, as much macroplastics as MPs). This is clearly less than the almost three OM difference we find between previous mass estimates of riverine MP fluxes and ours.

The smallest MPs at the lower micrometer scale and nanoplastics also contribute to oceanic stocks and river fluxes. Recent studies (18, 22) highlighted that these particles could represent a significant or even dominant part in the oceanic plastic stocks (mainly below surface). Our data confirm their abundance in rivers, at least in terms of number concentrations. However, this should have no impact on our calculations of residence times as we only refer to MP debris collected by plankton net sampling, which do not retain such small particles.

### 3.1.2 Other terrestrial MP sources

Beyond rivers, other pathways involved in the land-to-sea transfer of plastics are atmospheric transport and deposition, direct wastewater inputs from coastal cities, settlements and the areas nearby, and direct dumping from the shoreline. Unfortunately, all these sources are poorly quantified at global scale. Would they be quantitatively much greater than MP discharge from rivers, the residence time of MPs at the ocean surface would indeed decrease much further. But OM calculations indicate that these sources are clearly less important than rivers. Besides the smallest MPs and nanoplastics, atmospheric MP deposits on land correspond almost exclusively to fibers (38, 62, 63). Consequently, fibers should also be the dominant MP form from atmospheric deposition at sea. It could be eventually assumed that atmospheric fiber deposits on land are integrally transferred to the sea by rivers via surface runoff. However, this would lead to an overestimation as fibers can also originate from other sources such as wastewaters. The average mass percentage of fibers in our “Plankton net – River” subset of data in Table 1 is 1.1% (n=68 when omitting the case studies in which fibers were not measured), which corresponds to a global mass flux of 0.08 kt yr<sup>-1</sup> of MPs. Assuming an equal deposition rate over the entire ocean, i.e. over an area about 3 times larger than the land area draining into the sea, atmospheric fiber deposits at sea should therefore be <0.25 kt yr<sup>-1</sup>. Again, smallest fibers and MPs may not be retained in plankton nets and actual atmospheric deposition rates could be greater. But as we compare plankton net sampling in rivers with plankton net sampling at the sea surface, this should have no impact on the average residence times we calculate. Very recently, a study detected airborne MP particles in the marine atmosphere over the Atlantic, which were all clearly smaller than 300 µm in size (64).

To our best knowledge, only one global modeling study (65), although poorly constrained by observations, quantified MP discharges to the sea by wastewaters. The authors estimated them to be 9.4 kt yr<sup>-1</sup>, i.e. in the same range we found for riverine MPs in general. In their study, all wastewater inputs are supposed to enter the oceans via rivers, hence being included in our estimate. However, in coastal areas, wastewaters may be directly discharged into the sea via submarine outfall systems, specially off large cities. According to our GIS, 18.9% of the world population lives in coastal areas within <50 km distance to the sea and could potentially be connected to such systems. Assuming approximately that wastewaters from up to one quarter of this population may be drained through submarine outfall systems would consequently produce an additional flux estimate of 0.44 kt yr<sup>-1</sup>. This is clearly lower than riverine MP loads as a whole.

### 3.1.3 Fragmentation

Fragmentation of larger plastic debris can also contribute to the floating MP stock at sea. Because of the difficulties in studying this process under field conditions, there is no estimate quantifying its significance at the global scale. However, comparison of the average MP particle masses both in rivers and at the ocean surface is indicative in this respect. If fragmentation happens rapidly, and rivers remain a dominant source, ocean MPs would be expected to be smaller on average than riverine MPs. If the opposite, small MPs of riverine origin could be preferentially withdrawn from the ocean surfaces as they would have lower buoyancies (33, 66) compared to large MPs. In that case, it would be the larger MPs that accumulate at the ocean surface.

The study of Cózar et al. (12) is of interest in this context because it allows estimating the representative standard mass of MP debris at the ocean surface from regression analysis. Based on the relationship between mass and number densities the authors published in their supplementary

data section, together with the average MP density derived from their mid-estimate MP stock (44.5 g km<sup>-2</sup>), we calculated that this mass is about 0.97 mg item<sup>-1</sup>. Regionally, it ranges from approximately 0.3 mg item<sup>-1</sup> in low-density (~10<sup>2</sup> items km<sup>-2</sup>) areas to 3.0 mg item<sup>-1</sup> in high-density (~10<sup>7</sup> items km<sup>-2</sup>) areas. Due to the log-log nature of the cited relationship, which implies normal data distribution, the standard MP mass in the ocean can be directly compared to our MP mass estimate. This consequently represents about 4 times the median mass of 0.23 mg item<sup>-1</sup> we retained for riverine non-fiber MPs (for this purpose fibers can be neglected since they were often not accounted for in marine samples). The studies of Eriksen et al. (13) and van Sebille et al. (11) solely reported average number and mass budgets according to which the average MP weight in the oceans is 4.6–7.3 mg item<sup>-1</sup>, i.e. 10–15 times our average value for non-fiber MPs in rivers (0.48 mg item<sup>-1</sup>, see Fig. S1A). In all cases, oceanic MPs are hence heavier on average than riverine MPs, which does not support the idea that plastic debris undergo rapid fragmentation into floating MPs when they arrive to the marine domain.

### 3.1.4 Surface ocean stocks

It is finally also important to mention that estimates for the average residence time of MPs at the ocean surface not only depend on annual river loads but also on the quality of estimates of the floating MP stock in the ocean. For quantification of the latter, we refer to the study of Cózar et al. (12) as a conservative low-end estimate. Larger stocks would consequently increase the calculated average residence time. Although Cózar et al. (12) did not cut off plastic debris >5mm in their data compilation (they used an upper size limit of 100 mm, see above), the mid-estimate plastic stock they propose only corresponds to 41% of the estimated stock of Eriksen et al. (13) for plastic debris below this size. The former study assembled a larger number of field observations than the latter and covered many low-density areas in the oceans. At the high-end of estimates, the oceanic MP stock can reach more than 10 times Cózar's et al. (12) stock and differ significantly in terms of spatial distributions according to a series of modeling studies that are summarized in van Sebille et al. (11). However, whereas these studies still increased the number of observations for calibration, all models involved accounted for dynamic MP interactions at the land-to-sea interface (i.e. prediction of the land-derived inputs and/or outputs via beaching along the coastlines), thus making them less suitable for independent comparisons with observations, as the load estimates they produce directly depend on the algorithms controlling the land-to-sea interactions.

### 3.2 Regression uncertainties

The multi-parametric regression models we determined using the specific prediction parameters sPop and sQ only yield moderate coefficients of determination for the prediction of specific MP fluxes (Supplementary Materials and Methods 2.2). Based on the standard error of both regressions (total fluxes ± standard error), the corresponding error intervals for the fluxes reported in our Table 1 stretch from 1.3 to 30.1 kt yr<sup>-1</sup> (n=96) and from 1.3 to 31.7 kt yr<sup>-1</sup> (n=84). Weak coefficients of determination and large error intervals are not surprising given the constraints under which the MP flux estimates had to be established. These constraints include standard per item mass conversions instead of size specific mass conversions, missing information on hydrological conditions and seasonal variations, and restriction of plankton net sampling to low discharge conditions. The global fluxes we calculated should consequently be considered as OM estimates. To quantify the confidence we can associate to these estimates, we further submitted our regressions to a bootstrap

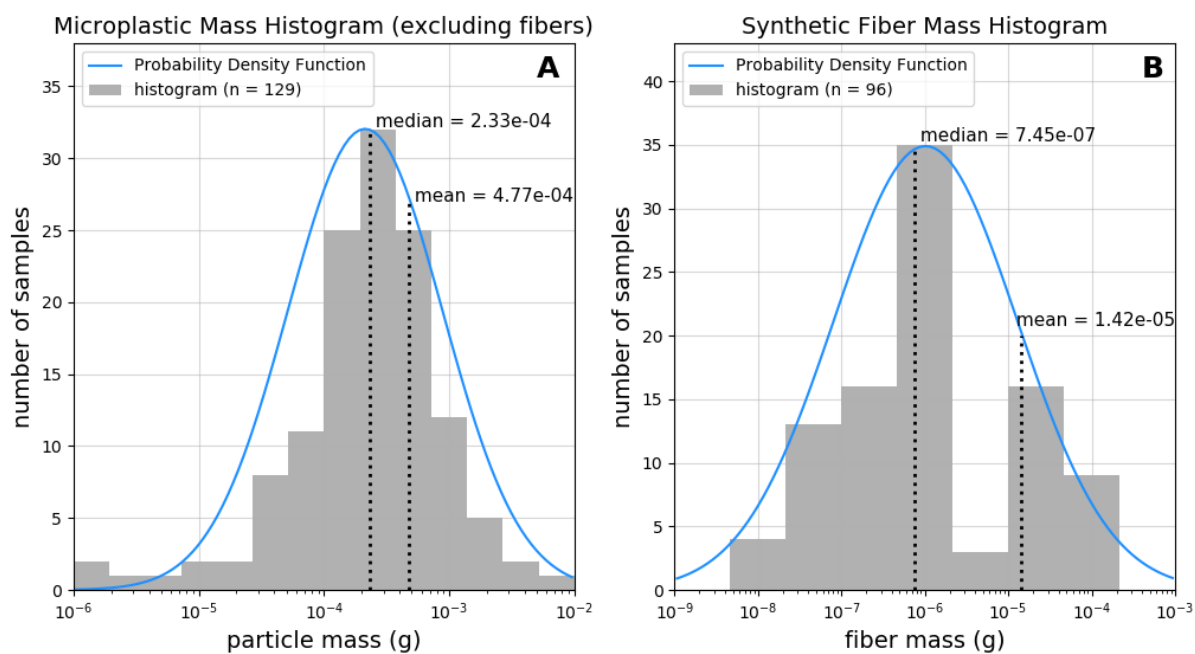
method (67) as it was done in Schmidt's et al. models (8). This method includes random data sampling from our data sets, the computation of associated models and MP fluxes (5,000 runs), and the determination of the 95% confidence intervals according to the improved percentiles (Bca) method. These intervals were calculated to 3.6–13.9 kt yr<sup>-1</sup> (n=96) and to 3.4–12.6 kt yr<sup>-1</sup> (n=84) and in both cases they encompass our global fluxes reported in Table 1.

## SUPPLEMENTARY REFERENCES

47. C. J. Moore, G. L. Lattin, A. F. Zellers, Density of Plastic Particles found in zooplankton trawls from Coastal Waters of California to the North Pacific Central Gyre. Proceedings of the Plastic Debris Rivers to Sea Conference, Algalita Marine Research Foundation, Long Beach, CA USA (2005).
48. A. L. Andrady, Plastics and the environment. In: Anthony L. Andrady (Ed.), Publisher: John Wiley & Sons (2004).
49. C. J. Moore, G. L. Lattin, A. F. Zellers, Quantity and type of plastic debris flowing from two urban rivers to coastal waters and beaches of Southern California. *J. Integr. Coast. Zone Manag.* 11, 65–73 (2011).
50. S. S. Sadri, R. C. Thompson, On the quantity and composition of floating plastic debris entering and leaving the Tamar Estuary, Southwest England. *Mar. Pollut. Bull.* 81, 55-60 (2014).
51. J. Gasperi, R. Dris, T. Bonin, V. Rocher, B. Tassin, Assessment of floating plastic debris in surface water along the Seine River. *Environ. Pollut.* 195, 163-166 (2014).
52. M. Constant, P. Kerhervé, M. Mino-Vercellio-Verollet, M. Dumontier, A. Sánchez-Vidal, M. Canals, S. Heussner, Beached microplastics in the northwestern Mediterranean Sea. *Mar. Pollut. Bull.* 142, 263–273 (2019).
53. L. T. Yonkos, E. A. Friedel, A. C. Perez-Reyes, S. Ghosal, C. D. Arthur, Microplastics in four estuarine rivers in the Chesapeake Bay, USA. *Environ. Sci. Technol.* 48, 14195-14202 (2014).
54. M. van der Wal, M. van der Meulen, G. Tweehuijsen, M. Peterlin, A. Palatinus, M. K. Virsek, L. Coscia, A. Krzan, Identification and Assessment of Riverine Input of (Marine) Litter. Final Report for the European Commission DG Environment under Framework Contract No ENV.D.2/FRA/2012/0025 (2015).
55. M. O. Rodrigues, N. Abrantes, F. J. M. Goncalves, H. Nogueira, J. C. Marques, A. M. M. Goncalves, Spatial and temporal distribution of microplastics in water and sediments of a freshwater system (Antua River, Portugal). *Sci. Total Environ.* 633, 1549-1559 (2018).
56. L. Lahens, E. Strady, T. C. Kieu Le, R. Dris, B. Kada, E. Rinnert, J. Gasperi, B. Tassin, Macroplastic and microplastic contamination assessment of a tropical river (Saigon River, Vietnam) transversed by a developing megacity. *Environ. Pollut.* 236, 661-671 (2018).
57. T. Kataoka, Y. Nihei, K. Kudou, H. Hinata, Assessment of the sources and inflow processes of microplastics in the river environments of Japan. *Environ. Pollut.* 244, 958–965 (2018).
58. Center for International Earth Science Information Network - CIESIN - Columbia University. 2018. 30 arc-second resolution Gridded Population of the World for the year 2015, Version 4 (GPWv4): Population Density, Revision 11. Palisades, NY: NASA Socioeconomic Data and Applications Center (SEDAC). <https://doi.org/10.7927/H49C6VHW> (2018).
59. W. Ludwig, P. Amiotte-Suchet, G. Munhoven, J. L. Probst, Atmospheric CO<sub>2</sub> consumption by continental erosion: Present-day controls and implications for the last glacial maximum. *Glob. Planet. Change* 16, 95-108 (1998a).
60. W. Ludwig, J. L. Probst, River sediment discharge to the oceans: Present-day controls and global budgets. *American J. Sci.* 296, 265-295 (1998b).
61. P. Döll, B. Lehner, Validation of a new global 30-min drainage direction map. *J. Hydrol.* 258, 214–231 (2002).
62. R. Dris, J. Gasperi, M. Saad, C. Mirande, B. Tassin, Synthetic fibers in atmospheric fallout: A source of microplastics in the environment? *Mar. Pollut. Bull.* 104, 290-293 (2016).
63. L. Cai, J. Wang, J. Peng, Z. Tan, Z. Zhan, X. Tan, Q. Chen, Characteristic of microplastics in the atmospheric fallout from Dongguan city, China: Preliminary research and first evidence. *Environ. Sci. Pollut. Res.* 24, 24928-24935 (2017).
64. M. Trainic, J. M. Flores, I. Pinkas, M. L. Pedrotti, F. Lombard, G. Bourdin, G. Gorsky, E. Boss, Y. Rudich, A. Vardi, I. Koren, Airborne microplastic particles detected in the remote marine atmosphere. *Commun. Earth Environ.* 1, 64 (2020).

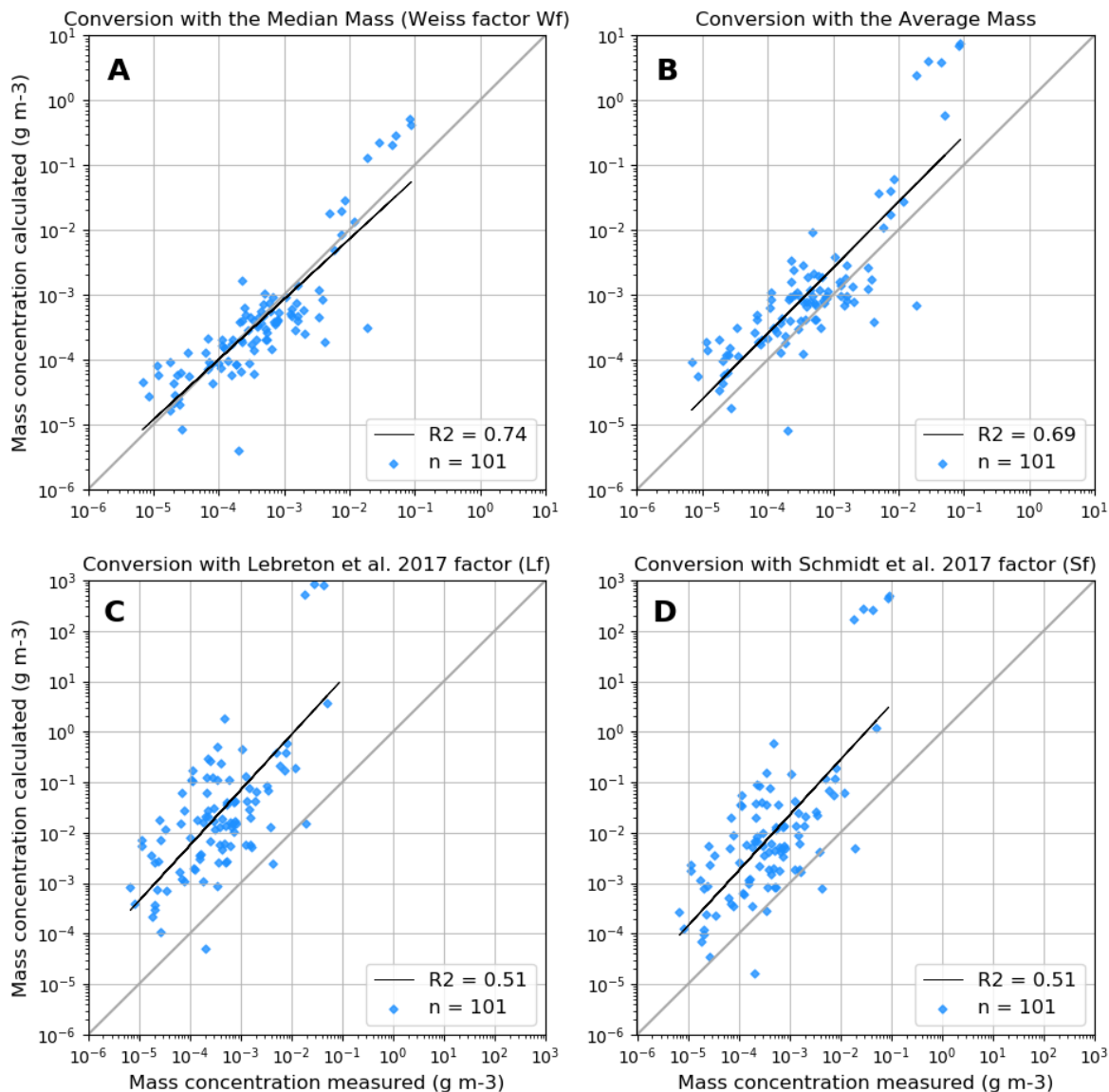
65. J. van Wijnen, A. M. Ragas, C. Kroeze, Modelling global river export of microplastics to the marine environment: Sources and future trends. *Sci Total Environ.* 673, 392-401 (2019).
66. M. Kooi, J. Reisser, B. Slat, F. F. Ferrari, M. S. Schmid, S. Cunsolo, R. Brambini, K. Noble, L.-A. Sirks, T. E. W. Linders, R. I. Schoeneich-Argent, A. A. Koelmans, The effect of particle properties on the depth profile of buoyant plastics in the ocean. *Sci. Rep.* 6, 1-10 (2016).
67. D. Kushary, Bootstrap methods and their application. *Technometrics* 42, 216-217 (2000).
68. A. K. Baldwin, S. R. Corsi, S. A. Mason, Plastic Debris in 29 Great Lakes Tributaries: Relations to Watershed Attributes and Hydrology. *Environ. Sci. Technol.* 50, 10377–10385 (2016).
69. T. Mani, A. Hauk, U. Walter, P. Burkhardt-Holm, Micro-plastics profile along the Rhine River. *Sci. Rep.* 5, 17988 (2016).
70. W. Wang, A. W. Ndungu, Z. Li, J. Wang, Microplastics pollution in inland freshwaters of China: A case study in urban surface waters of Wuhan, China. *Sci. Total Environ.* 575, 1369–1374 (2017).
71. A. R. McCormick, T. J. Hoellein, S. A. Mason, J. Schlupe, J. J. Kelly, Microplastic is an Abundant and Distinct Microbial Habitat in an Urban River. *Environ. Sci. Technol.* 48, 11863–11871 (2014).
72. A. R. McCormick, T. J. Hoellein, M. G. London, J. Hittie, J. W. Scott, J. J. Kelly, Microplastic in surface waters of urban rivers: concentration, sources, and associated bacterial assemblages. *Ecosphere* 7, e01556 (2016).
73. S. Rech, V. Macaya-Caquilpan, J. F. Pantoja, M. M. Rivadeneira, C. K. Campodonico, M. Thiel, Sampling of riverine litter with citizen scientists - findings and recommendations. *Environ. Monit. Assess.* 187, (2015).
74. A. Vianello, F. Acri, F. B. Aubry, A. Boldrin, E. Camatti, L. Da Ros, T. Marceta, V. Moschino, Occurrence and distribution of floating microplastics in the North Adriatic Sea: preliminary results. *Proceedings of the MICRO2015 Seminar on microplastics (Piran, 2015)*.
75. S. Estahbanati, N. L. Fahrenfeld, Influence of wastewater treatment plant discharges on microplastic concentration in surface water. *Chemosphere* 162, 277-284 (2016).
76. A. R. A. Lima, M. F. Costa, M. Barletta, Distribution patterns of microplastics within the plankton of tropical estuary. *Environ. Res.* 132, 146-155 (2014).
77. P. Hohenblum, B. Liebmann, M. Liedermann, Plastic and microplastic in the environment. *Umweltbundesamt REP-0551, Vienna* (2015).
78. J. C. Vermaire, C. Pomeroy, S. M. Herczegh, O. Haggart, M. Murphy, Microplastic abundance and distribution in the open water and sediment of the Ottawa River, Canada, and its tributaries. *FACETS* 2, 301–314 (2017).
79. R. Z. Miller, A. J. R. Watts, B. O. Winslow, T. S. Galloway, A. P. W. Barrows, Mountains to the sea: River study of plastic and non-plastic microfiber pollution in the northeast USA. *Mar. Pollut. Bull.* (2017).
80. S. H. Campbell, P. R. Williamson, B. D. Hall, Microplastics in the gastrointestinal tracts of fish and the water from an urban prairie creek. *FACETS* 2, 395–409 (2017).
81. L. K. Schmidt, M. Bochow, H. K. Imhof, S. E. Oswald, Multi-temporal surveys for microplastic particles enabled by a novel and fast application of SWIR imaging spectroscopy - Study of an urban watercourse traversing the city of Berlin, Germany. *Environ. Pollut.* 239, 579-589 (2018).
82. L. Lin, L. Z. Zuo, J. P. Peng, L. Q. Cai, L. Fok, Y. Yan, H. X. Li, X. R. Xu, Occurrence and distribution of microplastics in an urban river: A case study in the Pearl River along Guangzhou City, China. *Sci. Total Environ.* 644, 375-381 (2018).
83. K. J. Kapp, E. Yeatman, Microplastic hotspots in the Snake and Lower Columbia rivers: A journey from the greater Yellowstone ecosystem to the Pacific Ocean. *Environ. Pollut.* 241, 1082–1090 (2018).
84. R. Dris, J. Gasperi, V. Rocher, B. Tassin, Synthetic and non-synthetic anthropogenic fibers in a river under the impact of Paris Megacity: sampling methodological aspects and flux estimations. *Sci. Total Environ.* 618, 157-164 (2018).
85. A. P. W. Barrows, K. S. Christiansen, E. T. Bode, T. J. Hoellein, A watershed- scale, citizen science approach to quantifying microplastic concentration in a mixed land-use river. *Water Res.* 147, 382–392 (2018).
86. M. Yan, H. Nie, K. Xu, Y. He, Y. Hu, Y. Huang, J. Wang, Microplastic abundance, distribution and composition in the Pearl River along Guangzhou city and Pearl River estuary, China. *Chemosphere* 217, 879-886 (2019).
87. L. Watkins, S. McGrattan, P. J. Sullivan, M. T. Walter, The effect of dams on river transport of microplastic pollution. *Sci. Total Environ.* 664, 834-840 (2019).
88. X. Tan, X. Yu, L. Cai, J. Wang, J. Peng, Microplastic and associated PAHs in surface water from the Feilaixia Reservoir in the Beiji River, China. *Chemosphere* 221, 834–840 (2019).
89. L. Simon-Sánchez, M. Gerlaud, J. Garcia-Orellana, P. Ziveri, River Deltas as hotspots of microplastic accumulation: The case study of the Ebro River (NW Mediterranean). *Sci. Total Environ.* 687, 1186-1196 (2019).
90. A. D. Gray, H. Wertz, R. R. Leabs, J. E. Weinstein, Microplastic in two South Carolina Estuaries: Occurrence, distribution, and composition. *Mar. Pollut. Bull.* 128, 223-233 (2018).

91. Z. T. Anderson, A. B. Cundy, I. W. Croudace, P. E. Warwick, O. Celis-Hernandez, J. L. Stead, A rapid method for assessing the accumulation of microplastics in the sea surface microlayer (SML) of estuarine systems. *Scientific Reports* 8, 9428 (2018).
92. W. Luo, L. Su, N. J. Craig, F. Du, C. Wu, H. Shi, Comparison of microplastic pollution in different water bodies from urban creeks to coastal waters. *Environ. Pollut.* 246, 174-182 (2018).
93. C. Jiang, L. Yin, Z. Li, X. Wen, X. Luo, S. Hu, H. Yang, Y. Long, B. Deng, L. Huang, Y. Liu, Microplastic pollution in the rivers of the Tibet Plateau. *Environ. Pollut.* 249, 91-98 (2019).
94. L. Ding, R. fan Mao, X. Guo, X. Yang, Q. Zhang, C. Yang, Microplastics in surface waters and sediments of the Wei River, in the northwest of China. *Sci. Total Environ.* 667, 427-434 (2019).
95. C. C. Deocaris, J. O. Allosada, L. T. Ardiente, L. G. G. Bitang, C. L. Dulohan, J. K. I. Lapuz, L. M. Padilla, V. P. Ramos, J. B. P. Padolina, Occurrence of microplastic fragments in the Pasig River. *H2O Open J.* 2, 92-100 (2019).
96. F. C. Alam, E. Sembiring, B. S. Muntalif, V. Suendo, Microplastic distribution in surface water and sediment river around slum and industrial area (case study: Ciwalengke River, Majalaya district, Indonesia). *Chemosphere* 224, 637-645 (2019).

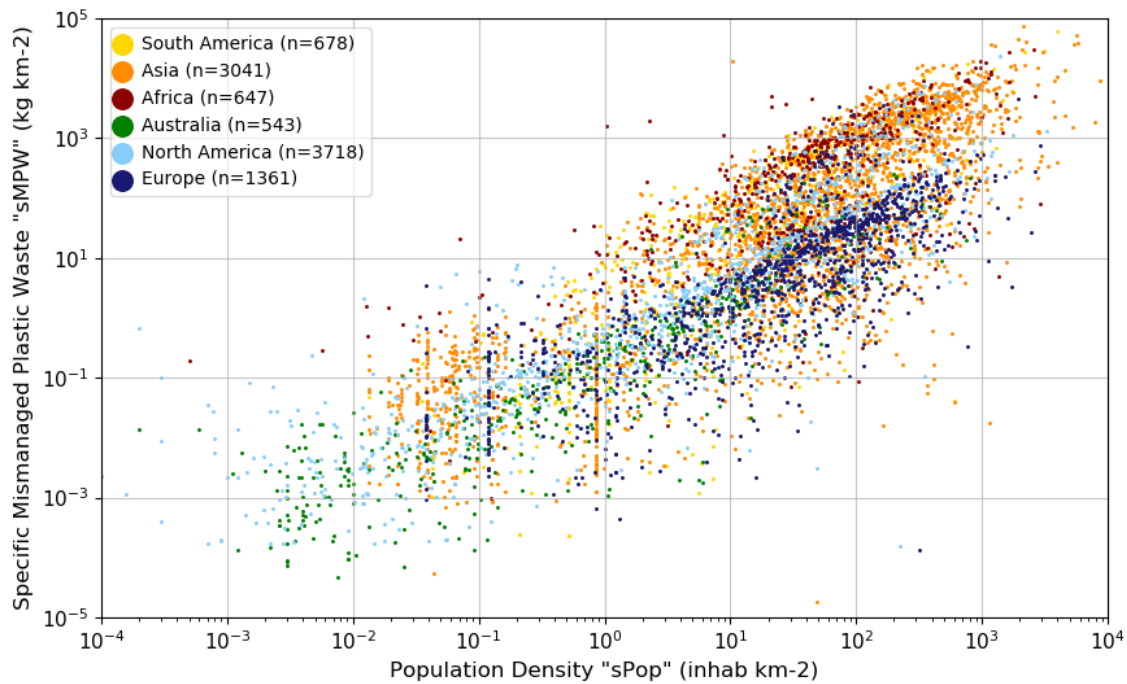


**Fig. S1. Mass histograms and probability density functions.** (A) For non-fiber MPs and (B) for fiber MPs. Both were calculated from literature studies in Table S1 in which both mass and number concentrations have been determined (see Supplementary Materials and Methods 1.3).





**Fig. S2. Comparison between measured and calculated MP mass concentrations.** Determined from (A) the combination of median masses of fiber and non-fiber MPs ( $W_f$ ) in Fig. S1, (B) the combination of average masses of fiber and non-fiber MPs in Fig. S1, (C) the average MP mass proposed by Lebreton et al. (7) ( $L_f$ ), and (D) the average MP mass proposed by Schmidt et al. (8) ( $S_f$ ).



**Fig. S3. Comparison between specific Mismanged Plastic Waste (MPW) generation (5) and 2015 population densities (58) for each exorheic basin in our global GIS (0.5 x 0.5 degree grid cell resolution). sMPW is about two orders of magnitude (OM) more variable than sPop, which can explain why MPW-based extrapolations generally lead to much greater budgets than extrapolations with the Population. Specific MPW generation rates are generally higher in African, South American and Asian countries than in European and North American countries.**

**Table S1. Compiled literature studies providing MP concentrations in rivers.** For data categorization as “plankton net sampling” and “river-like” see Supplementary Materials and Methods section 1.2 and for details on the calculation of MP mass concentrations see section 1.3.

ID	Reference	Sampling method and mesh size	River / Sampling point	Mean particle concentration (item m <sup>-3</sup> )	Mean fiber concentration (item m <sup>-3</sup> )	Mass concentration measured (g m <sup>-3</sup> )	Mass concentration calculated (g m <sup>-3</sup> )	River like	Plankton net
1			St. Louis River at Scanlon	7.48E-01	6.88E-01		1.45E-05	1	1
2			Nemadji River near South Superior	1.84E+00	1.76E+00		2.00E-05	1	1
3			Manitowoc River at Manitowoc	7.05E-01	6.20E-01		2.03E-05	1	1
4			Sheboygan River at Sheboygan	9.88E-01	8.93E-01		2.28E-05	1	1
5			Milwaukee River at Milwaukee	6.77E+00	1.06E+00		1.33E-03	1	1
6			Indiana Harbor Canal at E. Chicago	7.04E+00	1.23E+00		1.36E-03	0	1
7			Burns Ditch at Portage	3.10E-01	5.33E-02		5.99E-05	0	1
8			St. Joseph River at Niles	3.51E+00	4.40E-01		7.17E-04	1	1
9			Paw Paw River at Riverside	5.44E+00	5.25E+00		4.82E-05	1	1
10			Kalamazoo River at New Richmond	1.89E+00	1.57E+00		7.47E-05	1	1
11			Grand River near East Manville	3.58E+00	3.14E+00		1.04E-04	1	1
12			Saginaw River at Saginaw	4.76E+00	4.23E+00		1.27E-04	1	1
13			Clinton River at Moravian Dr.	1.22E+01	4.57E+00		1.79E-03	1	1
14			River Rouge at Detroit	9.81E+00	6.32E+00		8.19E-04	1	1
15	Baldwin et al (68)	Drifnet 0.333 µm	Huron River at Ann Arbor	8.83E+00	3.06E+00		1.35E-03	1	1
16			River Raisin near Monroe	1.38E+00	1.18E+00		4.64E-05	1	1
17			Maumee River at Waterville	2.43E+00	2.26E+00		3.96E-05	1	1
18			Portage River near Woodville	1.65E+00	1.53E+00		2.91E-05	1	1
19			Sandusky River near Fremont	3.02E+00	2.34E+00		1.60E-04	1	1
20			Huron River at Milan	1.07E+00	9.33E-01		3.22E-05	1	1
21			Black River at Elyria	1.19E+00	8.10E-01		9.00E-05	1	1
22			Rocky River near Berea	2.86E+00	2.29E+00		1.35E-04	1	1
23			Cuyahoga River at Independence	3.55E+00	2.97E+00		1.38E-04	1	1
24			Grand River near Painesville	1.71E+00	1.18E+00		1.25E-04	1	1
25			Ashtabula River at Ashtabula	1.28E+01	1.17E+01		2.69E-04	1	1
26			Buffalo River at Michigan Ave	1.18E+01	4.12E+00		1.80E-03	1	1
27			Tonawanda Creek at Rapids	3.43E+00	3.17E+00		6.36E-05	1	1
28			Genesee River at Ford St	4.15E+00	3.34E+00		1.93E-04	1	1
29			Fox River at Oil Tank Depot	3.82E+00	2.22E+00		3.75E-04	1	1
30	Lechner et al (40)	Stationary conical driftnets 0.5 µm	Danube (Vienna – Bratislava)	4.13E-01	0.00E+00	2.20E-03	9.63E-05	1	1
31			Seine M1	2.80E-01	1.19E-01		3.77E-05	1	1
32	Dris et al (31)	Manta trawl 0.330 µm	Seine M2	2.91E-01	1.24E-01		3.91E-05	1	1
33			Seine M3	4.72E-01	2.01E-01		6.34E-05	1	1
34			Rhône upstream	2.30E+00	2.30E-01	2.60E-04	4.83E-04	1	1
35			Aubonne	1.00E-01	1.00E-02	2.00E-05	2.10E-05	1	1
36			Venoge	3.53E+01	3.53E+00	6.80E-03	7.41E-03	1	1
37	Faure et al (39)	Manta trawl 300 µm	Vuachère	4.40E+00	4.40E-01	6.70E-04	9.24E-04	1	1
38			Rhône Geneva	1.30E-01	1.30E-02	8.40E-06	2.73E-05	1	1
39			Rhône Chancy	2.90E-01	2.90E-02	3.50E-04	6.09E-05	1	1
40			Rhine Basel	1.32E+00	3.31E-02		3.01E-04	1	1
41			Rhine Strsb/Kehl	1.51E+00	3.76E-02		3.42E-04	1	1
42			Rhine Seltz	7.98E-01	1.99E-02		1.82E-04	1	1
43			Rhine Mainz	8.38E-01	2.10E-02		1.91E-04	1	1
44			Rhine Bad Honnef	3.27E+00	8.18E-02		7.45E-04	1	1
45	Mani et al (69)	Manta net 300 µm	Rhine Cologne Porz	3.77E+00	9.43E-02		8.58E-04	1	1
46			Rhine Leverkusen	4.81E+00	1.20E-01		1.09E-03	1	1
47			Rhine Duisburg	1.38E+01	3.45E-01		3.14E-03	1	1
48			Rhine Rees	1.71E+01	4.28E-01		3.89E-03	1	1
49			Rhine Zuilichem	5.71E+00	1.43E-01		1.30E-03	1	1
50			Rhine Rotterdam	1.60E+00	3.99E-02		3.63E-04	0	1
51	Wang et al (70)	Teflon pump 0-20 cm depth + 50 µm stainless steel sieve	Yangtze river (11 sites)	2.52E+03	1.85E+03		1.58E-01	1	0
52			Hanjiang river (5 sites)	2.93E+03	2.18E+03		1.78E-01	1	0
53	McCormick (71)		North Shore Channel	9.94E+00	5.89E+00		9.48E-04	0	1
54			Higgen's Cr. Upstream WWTP	5.90E+00	1.07E+00		1.13E-03	1	1
55			Springbrook Creek	3.28E+00	1.16E+00		4.97E-04	1	1
56			L Kickapoo Creek	1.02E+00	6.45E-01		8.80E-05	1	1
57	McCormick et al (72)	Neuston net 333 µm	N. Shore Channel	4.98E+00	2.34E+00		6.19E-04	0	1
58			Goose Creek	3.45E+00	1.58E+00		4.39E-04	1	1
59			DuPage River	8.10E+00	3.96E+00		9.70E-04	1	1
60			W Br DuPage River	1.95E+00	9.10E-01		2.42E-04	1	1
61			Salt Creek	2.11E+00	1.13E+00		2.28E-04	1	1
62			E Br DuPage River	6.00E+00	3.59E+00		5.65E-04	1	1
63			Elqui	1.16E-01	0.00E+00		2.72E-05	1	1
64			Maipo	1.15E+00	0.00E+00		2.69E-04	1	1
65	Rech et al (73)	Neuston net 1 mm	Maule mouth	7.40E-01	0.00E+00		1.73E-04	1	1
66			BioBio mouth	4.83E-02	0.00E+00		1.13E-05	1	1

**Table S1 (continued). Compiled literature studies providing MP concentrations in rivers.** For data categorization as “plankton net sampling” and “river-like” see Supplementary Materials and Methods section 1.2 and for details on the calculation of mass concentrations see section 1.3.

ID	Reference	Sampling method and mesh size	River / Sampling point	Mean particle concentration (item m <sup>-3</sup> )	Mean fiber concentration (item m <sup>-3</sup> )	Mass concentration measured (g m <sup>-3</sup> )	Mass concentration calculated (g m <sup>-3</sup> )	River like	Plankton net
67			Coyote Creek (channelized part)	1.24E+01	-		2.89E-03	0	0
68	Moore et al (49)	Manta trawl, Streambed, Handnets 333 to 800 µm	San Gabriel river (channelized part)	4.59E+01	-		1.07E-02	0	0
69			Los Angeles river (channelized part)	1.44E+03	-		3.37E-01	0	0
70			Rhine 1	4.92E+00	4.92E-02		1.14E-03	1	1
71			Rhine 2	1.85E+00	2.78E-01		3.67E-04	1	1
72	Van der Wal et al (54)	Manta net 330 µm	Po	1.46E+01	2.19E+00		2.90E-03	1	1
73			Danube	1.06E+01	7.42E+00		7.48E-04	1	1
74			Dalalven	4.54E+00	4.54E-02		1.05E-03	1	1
75	Vianello et al (74)	Manta trawl 330 µm	Po mouth (seaward side)	6.60E+00	0.00E+00		1.54E-03	0	1
76	Zhao et al (34)	Teflon pump 1m depth sieve 32µm	Yangtze estuary	4.14E+03	3.27E+03		2.04E-01	0	0
77			Patapsco tidal river (bay)	1.04E+00	0.00E+00	6.79E-04	2.42E-04	0	1
78			Magothy tidal river (bay)	7.51E-01	0.00E+00	4.94E-04	1.75E-04	0	1
79	Yonkos et al (53)	Manta net 330 µm	Rhode tidal river (bay)	4.50E-01	0.00E+00	1.20E-04	1.05E-04	0	1
80			Corsica tidal river (bay)	2.72E-01	0.00E+00	6.25E-05	6.35E-05	0	1
81			Sao Joao da Madeira (Antua river)	6.96E+02	4.30E+01	2.84E-02	1.52E-01	1	0
82	Rodrigues et al (55)	Pump (1.2 m3) + Net 55 µm	Aguineira (Antua river)	1.60E+02	5.70E+01	8.00E-03	2.41E-02	1	0
83			Estareja (Antua river)	6.40E+01	3.55E+01	6.65E-03	6.68E-03	1	0
84	Estahbanati et al (75)	Net 153 µm	Raritan River	4.79E+01	-		9.96E-03	1	1
85	Lima et al (76)	Plankton net 300 µm	Goiana river	2.61E-01	3.64E-02		5.23E-05	1	1
86	Sadri et al (50)	Manta net 300 µm	Tamar tidal estuary	2.80E-02	8.00E-04		6.35E-06	0	1
87	Lahens et al (56)	Net 300 µm	Saigon River	1.76E+02	0.00E+00	1.50E-04	4.11E-02	1	1
88	Hohenblum et al (77)	Drift net 500 µm (rarely 250 µm)	Danube Hainburg	3.80E-01	0.00E+00	1.06E-04	8.86E-05	1	1
89			Danube Aschach	1.23E-01	0.00E+00	1.33E-05	2.86E-05	1	1
90	Constant et al (38)	Manta and conical nets 333 µm	Rhône	1.54E+01	1.46E+01	3.24E-04	2.16E-04	1	1
91			Têt	4.23E+01	4.05E+01	8.00E-04	4.43E-04	1	1
92	Vermaire et al (78)	Manta trawl 100 µm	Ottawa river (8 sites)	1.35E+00	9.86E-01		8.58E-05	1	1
93	Miller et al (79)	Bucket 3L, filter 45µm for fibers	Hudson river (142 sites)	9.80E+02	9.80E+02		7.31E-04	1	0
94	Campbell et al (80)	Conical net 80 µm	Wascana creek	3.02E+00	2.70E+00		7.60E-05	1	1
95	Schmidt et al (81)	Niskin bottle 2.2L, Net 20 µm	Teltow artificial canal Berlin	7.86E+03	-		1.83E+00	0	0
96	Lin et al (82)	Bulks 5L 50 cm depth, 20µm sieve	Pearl river (Guangzhou)	2.72E+03	2.20E+03		1.23E-01	1	0
97	Kapp et al (83)	Plankton net 100µm + Glas jar 2L for fibers	Snake river (tributary)	2.45E+00	7.78E-01		3.90E-04	1	1
98			Columbia river	2.97E+00	2.60E+00		8.69E-05	1	1
99			Seine P1	1.01E+02	1.01E+02		7.50E-05	1	1
100			Seine P2	4.85E+01	4.85E+01		3.62E-05	1	1
101	Dris et al (84)	Plankton net 80 µm for microfibers only	Seine P3	2.79E+01	2.79E+01		2.08E-05	1	1
102			Seine P4	2.79E+01	2.79E+01		2.08E-05	1	1
103			Seine P5	2.21E+01	2.21E+01		1.65E-05	1	1
104	Barrows et al (85)	Bottles 1L, filter 100 µm	Gallatin river (72 sites)	1.30E+03	1.04E+03		6.14E-02	1	0
105			Pearl river (Guangzhou)	1.99E+04	1.79E+03		4.31E+00	1	0
106	Yan et al (86)	Bulks 5L + 50 µm stainless steel sieve	Pearl river tidal estuary	8.90E+03	8.01E+02		1.90E+00	0	0
107	Xiong et al (35)	Net 333 µm	Yangtze river	9.00E-01	7.11E-02		1.93E-04	1	1
108	Watkins et al (87)	Plastic bottle 335µm sieve	Fall creek dams	1.58E+01	1.42E+01		3.79E-04	0	0
109			Six mile creek dams	5.00E+00	4.50E+00		1.20E-04	0	0
110	Tan et al (88)	Plankton net 112 µm	Beijiang river (feilaixia resevoir)	5.60E-01	8.81E-02		1.10E-04	0	1
111	Simon-Sánchez et al (89)	Neuston net 5 µm	Ebro river	3.50E+00	1.61E+00		4.42E-04	1	1
112	Gray et al (90)	Sea surface layer apparatus, 2 mm sieve	Charleston harbor	6.60E+03	3.70E+03		6.80E-01	0	0
113			Winyah bay	3.08E+04	1.05E+03		6.94E+00	0	0
114	Anderson et al (91)	Glass plate, 1.2mm sieve, bulks 2L, filter 45µm for fibers	Hamble tidal estuary	1.72E+04	1.72E+04		1.28E-02	0	0
115			Beaulieu tidal estuary	7.05E+03	7.05E+03		5.26E-03	0	0
116	Luo et al (92)	Metal pail 5L, air lift pump, filter 20 µm	Yangtze estuary	9.00E+02	5.94E+02		7.18E-02	0	0
117			Huangpu and Suzhou rivers	2.08E+03	1.76E+03		7.64E-02	1	0
118	Kataoka et al (57)	Plankton net 335 µm	29 Japan rivers	1.60E+00	0.00E+00	4.40E-04	3.73E-04	1	1
119			Buqu river	9.67E+02	6.67E+02		7.04E-02	1	0
120			Naqu river	8.17E+02	7.52E+02		1.58E-02	1	0
121	Jiang et al (93)	Large flow sampler + 45 µm stainless steel sieve	Lhasa river	6.92E+02	4.57E+02		5.51E-02	1	0
122			Brahmaputra river	4.83E+02	4.20E+02		1.50E-02	1	0
123			Nyang river	5.17E+02	4.08E+02		2.56E-02	1	0
124	Ding et al (94)	Pump 30 L, 75 µm sieve	Wei river	7.00E+00	3.51E+00		8.18E-04	1	0
125	Deocaris et al (95)	Manta trawl 355 µm	Pasig river	1.32E+00	0.00E+00		3.09E-04	1	1
126	Alam et al (96)	Glass container 1L + Filter 1.2 µm	Ciwalengke river (Majalaya)	5.85E+03	3.80E+03		4.81E-01	1	0
127	Lahens et al (56)	Bulk 300mL for fibers filter 2.7µm	Saigon River	3.42E+05	3.42E+05	3.17E-02	2.55E-01	1	0

**Table S2. Characteristics of the 0.5 x 0.5 degree resolution river basin GIS and integrated data layers used in this study.** Parentheses correspond to percentages of global values. For data sources, see Supplementary Materials and Methods 2.1.

	<b>Exorheic Basins</b>	<b>North Atlantic Ocean</b>	<b>South Atlantic Ocean</b>	<b>Indian Ocean</b>	<b>North Pacific Ocean</b>	<b>South Pacific Ocean</b>	<b>Other</b>
<i>Basins (number)</i>	9,988	2,526 (25)	422 (4)	1,028 (10)	1,895 (19)	1,091 (11)	3,026 (30)
<i>Area (M.km<sup>2</sup>)</i>	107.6	27.2 (25)	16.6 (15)	19.3 (18)	15.6 (15)	3.8 (4)	25.1 (23)
<i>Discharge (km<sup>3</sup> yr<sup>-1</sup>)</i>	44,600	14,945 (33)	5,791 (13)	5,425 (12)	9,318 (21)	4,334 (10)	4,847 (11)
<i>Population 2015 (billion inhabitants)</i>	6.99	0.97 (14)	0.72 (10)	2.20 (31)	2.08 (30)	0.28 (4)	0.74 (11)
<i>MPW - Jambeck et al., 2015 (Mt)</i>	109	5.8 (5)	11.3 (10)	21.1 (19)	56.2 (52)	4.3 (4)	10.3 (9)
<i>MPW - Lebreton et al., 2019 (Mt)</i>	64	3.2 (5)	10.4 (16)	21.6 (34)	20.3 (32)	1.0 (2)	7.5 (12)
<i>Suspended matter (Mt)</i>	15,640	3,603 (23)	509 (3)	3,631 (23)	5,372 (34)	1,614 (10)	911 (6)

**Table S3. Regression and correlation coefficients for the multi-regression models performed in this study.** All parameters are significant with  $p < 0.03$ . sPop: population density; sQ: drainage intensity. For the parametric equation, see Supplementary Materials and Methods 2.2.

<b>Data subsets</b>	<b>Number of data</b>	<b>Regression parameters</b>	<b>Number to mass conversion factor</b>	<b>r<sup>2</sup></b>	<b>Global MP flux (kt yr<sup>-1</sup>)</b>	<b>a1</b>	<b>a2</b>	<b>b</b>
<i>Plankton net</i>	96	sPop, sQ	Wf	0.35	6.1	3.39e-1	1.32	5.09e-6
<i>Plankton net - River</i>	84	sPop, sQ	Wf	0.38	6.6	3.31e-1	1.35	4.62e-6