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Research article

Assessing macroplastic debris collected from eight diverse river systems across four continents: Insights from synchronous three-year community-led research efforts

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A R T I C L E I N F O

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A B S T R A C T

Plastic pollution is an urgent and growing threat to human and planetary health. Rivers transport large volumes of plastic pollution across and between Earth's systems, providing opportunistic and strategic focal points for collection and quantitative assessments of plastic debris. A dearth of empirical, *in situ* studies of riverine plastic

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Plastic emissions
River plastic
Mismanaged plastic waste
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debris in scientific literature highlights the need for more research conducted in rivers across diverse contexts to better understand riverine plastic debris and inform upstream solutions. We present and analyze a dataset on macroplastic debris collected over three years (2020–2023) in a nearly continuous and synchronous fashion from eight diverse river systems in eight countries across four continents. We observed the majority (66 %) of the 3.8M kg of debris collected and analyzed in these river systems to be plastic. The compositions of polymers, single-use plastic items, and end-of-life fates of the collected plastic debris varied substantially between river systems. We discuss how differences in socioeconomic, regulatory, and infrastructure conditions across study sites begin to explain some of the observed variation. From these data insights, we share local and global recommendations for actions that could help reduce the flow of plastic debris into rivers in the first place. This research adds to our growing understanding of plastic pollution locally in these specific river systems as well as globally at a moment when the international community is actively working towards a global policy instrument to end plastic pollution.

1. Introduction

Plastic pollution is an urgent and growing threat to human and planetary health. Increases in plastic production coupled with insufficient capacity to manage the associated waste suggest that this problem will only grow in the decades ahead. Without intervention, global annual mismanaged plastic waste is predicted to double by 2050 (Pottinger, 2024). Resulting increases in plastic pollution have been predicted by myriad different data inputs and modelling approaches in recent years (Borrelle et al., 2020; Cottom et al., 2024; Lebreton and Andrady, 2019; Lau et al., 2020).

Rivers serve as a key transport vector of plastic debris across landscapes, between population centers, and from terrestrial to marine ecosystems (Schmidt et al., 2017; Lebreton et al., 2017; Mai et al., 2020; Nakayama and Osako, 2023; González-Fernández., 2021). Global annual riverine plastic emissions to the ocean have been estimated to range from 0.4 million metric tonnes (Mt) to 4 Mt (Lebreton, 2017; Meijer et al., 2021; Schmidt et al., 2017). Plastic debris in rivers and other aquatic and marine ecosystems has negative impacts on humans given the adjacency of communities to and their dependence upon these ecosystems. Inhalation and consumption of micro- and nanoplastic (much of which is the result of the degradation of macroplastic (van Wijnen et al., 2019a)) can lead to accumulation of these particles and associated chemical additives in human bodies (Qian et al., 2024; Smith et al., 2018; Amato-Lourenco, 2021; Kumar et al., 2020). There is yet much to learn about the resulting health impacts, but studies have demonstrated elevated risk for cardiovascular disease, inflammation, cancers, and reproductive harm (Wang and Qian, 2021; Trasande et al., 2018; Marfella et al., 2024; Weber et al., 2022; Zurub et al., 2024). In addition to direct impacts to human health, macroplastic can affect infrastructure by blocking drainages and exacerbating flood risk and damage, and has been shown to negatively influence socioeconomic activities such as tourism, fisheries, and shipping (McIlgorm et al., 2011; MacAfee and Löhr, 2024). Plastic debris in aquatic ecosystems also has negative impacts on biodiversity including direct impacts to wildlife via entanglement, ingestion, smothering, and leakage of chemical additives (Hahladakis et al., 2018; van Emmerik and Schwarz, 2020; Tian et al., 2021); transport of non-native species and pathogens (García-Gómez et al., 2021); and smothering and entanglement of mangrove forests and coral reefs (Tekman et al., 2022).

For the above reasons, rivers represent important focal systems for assessing plastic debris that contributes to plastic pollution, not only to understand rates and drivers of plastic emissions into and out of rivers, but also to shed light on the root sources from which this plastic pollution is generated in the first place. Past research on riverine plastic debris has communicated important patterns, such as that rivers are a primary vector for terrestrial plastic debris entering the marine environment (Lebreton and Andrady, 2019) and riverine plastic debris transport is highly variable on seasonal and interannual scales driven by weather and flood events (Axelsson et al., 2017; van Emmerik et al., 2019b). These results have been generated largely via modeling exercises (Schmidt et al., 2017; Lebreton et al., 2017; Mai et al., 2020; Meijer et al., 2021) and analyses of small-scale, localized plastic debris characterizations (van Emmerik et al., 2019b; Gasperi et al., 2014; Lechner et al., 2014; Morritt et al., 2014; Schirinzi et al., 2020) (with the exception of one study encompassing several rivers across Europe (González-Fernández., 2021)). The motivation for this research is underscored by the need for more empirical, *in situ* studies of riverine plastic debris, and especially more research conducted over longer time scales and synchronously in rivers across more diverse geographies to better understand the commonalities and differences of riverine plastic debris (González-Fernández et al., 2023; Roebroek et al., 2022).

The purpose of this research was to collect, analyze, and disseminate a novel dataset utilizing an equitable, community-led study design to help bridge the aforementioned knowledge gaps related to *in situ*, temporal, and spatial paucity of riverine macroplastic (>5 mm) data. We collected data on macroplastic debris for over three years (2020–2023) in a nearly continuous and synchronous fashion from eight river systems in eight countries (Mexico, Jamaica, Panama, Ecuador, Kenya, Vietnam, Thailand, and Indonesia) across four continents (Fig. 1, Table 1, Table S1). These rivers were selected to span a range of different watershed characteristics, and we focused on global south locations because there has traditionally been less investment, especially in community-based plastic pollution research projects (and therefore less data coverage), in these regions relative to rivers in the global north, which may bear equally significant levels of riverine plastic debris emissions. Collection of microplastic (<5 mm) was not feasible in the scope of this research.

The eight study sites encompassed a diverse range of geographic, hydrologic, cultural, and socioeconomic conditions, offering the opportunity to examine a variety of social, political, and environmental factors that influence the volume and composition of riverine macroplastic debris in different contexts (Fig. 1, Table S1). Macroplastic debris and associated data were collected in a coordinated fashion with local non-profit and social enterprise organizations at each study site as an intentional component of broader-scope river cleanup initiatives using methods tailored to the unique characteristics of each river (S1). While this community- and impact-centric approach to data collection can make fully orthogonal comparisons of patterns between study sites more challenging, it provided the advantage of directly involving local experts in the process of data collection and more equitably preserved the autonomy of local stakeholders designing and leading scientific research conducted in their own communities (Stefanoudis et al., 2021).

The contributions of this research are wide-ranging, as the novel empirical data on riverine macroplastic debris collected has implications from local to global levels and provides a large and unique dataset to a field of research lacking sufficient quantities of such types of information. Furthermore, we drew insights from this dataset to recommend actions that could help reduce the flow of plastic debris into rivers in the first place. Conclusions emerging from this research and future analyses of this dataset have local utility by contributing a more complete understanding of plastic debris in these specific river systems and insights into how to better manage it. These data can also be utilized for more large-scale applications, such as helping to calibrate global estimates of riverine plastic emissions generated using more theoretical, model-based approaches (S2.2). Additionally, these findings are timely as they improve our understanding of the range of riverine plastic debris characteristics across the globe in a moment where the international community is actively working towards a global policy instrument to end plastic pollution (UNEP, 2022).

2. Methods

Data on riverine macroplastic debris was collected between 2020 and 2023 from eight river systems in eight countries (Fig. 1, Table 1, Table S1). Data collection was performed in a coordinated fashion by local stakeholders at each study site as an intentional component of larger impact-driven river cleanup initiatives (S1). Study sites were selected to maximize the diversity of a variety of conditions in the dataset while also investing in plastic pollution cleanup and data collection programs in underserved communities to maximize real-world impact.

There was substantial variation in the size and flow characteristics of these eight rivers (Fig. 1, Table S1) ranging from small 7 m-wide ephemeral drainages that flowed only during the rainy season (i.e., Los Laureles Canyon, Mexico) to 800 m-wide perennial rivers (i.e., Red River, Vietnam). At six of the eight study sites (excluding Los Laureles Canyon, Mexico and Juan Díaz River, Panama), debris was collected at multiple locations in the local area (Table 1). The total number of months in which data were collected at the individual study sites ranged from 15 months in the Juan Díaz River, Panama to 43 months in the Athi River, Kenya (Table 1, Fig. 2). These discrepancies in sampling duration were driven by factors affecting the start date of collection such as variation in national or local policy for lockdowns during the COVID-19 pandemic and durations of project permitting and plastic debris collection technology construction periods. The study period for all sites ended on December 31, 2023.

To enable orthogonal comparisons in quantitative analyses across the study sites, we accounted for potential data biases introduced by study period variations and differences in collection totals, methods, effort, and seasonality across study sites (Table S3). The dataset was standardized as monthly means of total debris and plastic debris collected at each study site (Fig. 2) and proportions of specific characteristics of plastic debris collected at each study site relative to the totals of plastic debris collected respectively (Figs. 3–5).

2.1. *Riverine macroplastic debris data collection*

Four primary metrics were recorded by weight (kg) of macroplastic debris collected at each study site: 1) total debris (organic matter, metals, glass, plastic, and other materials), and as a subset, plastic debris, 2) major polymer classes (PET, HDPE, PVC, LDPE, PP, PS, Other), 3) select single-use plastic item categories (grocery and trash bags, beverage bottles, food wrappers, food and packaging foam) for a subset of the eight rivers (Athi River, Kenya; Citarum River, Indonesia; Portoviejo River, Ecuador; Red River, Vietnam), and 4) end-of-life fates for the collected plastic debris (recycled, waste-to-energy, downcycled, reused, landfilled). See Table 2 for descriptions of primary metrics. Microplastic (<5 mm) was not collected or reported in this study although it is important to acknowledge it was likely present in the studied river systems and the potential resultant impacts on environment, society, and health (Margenat et al., 2021; Kumar et al., 2021).

At each study site, all collected debris (organic, plastic, inorganic non-plastic) was removed from the river and weighed. Macroplastic

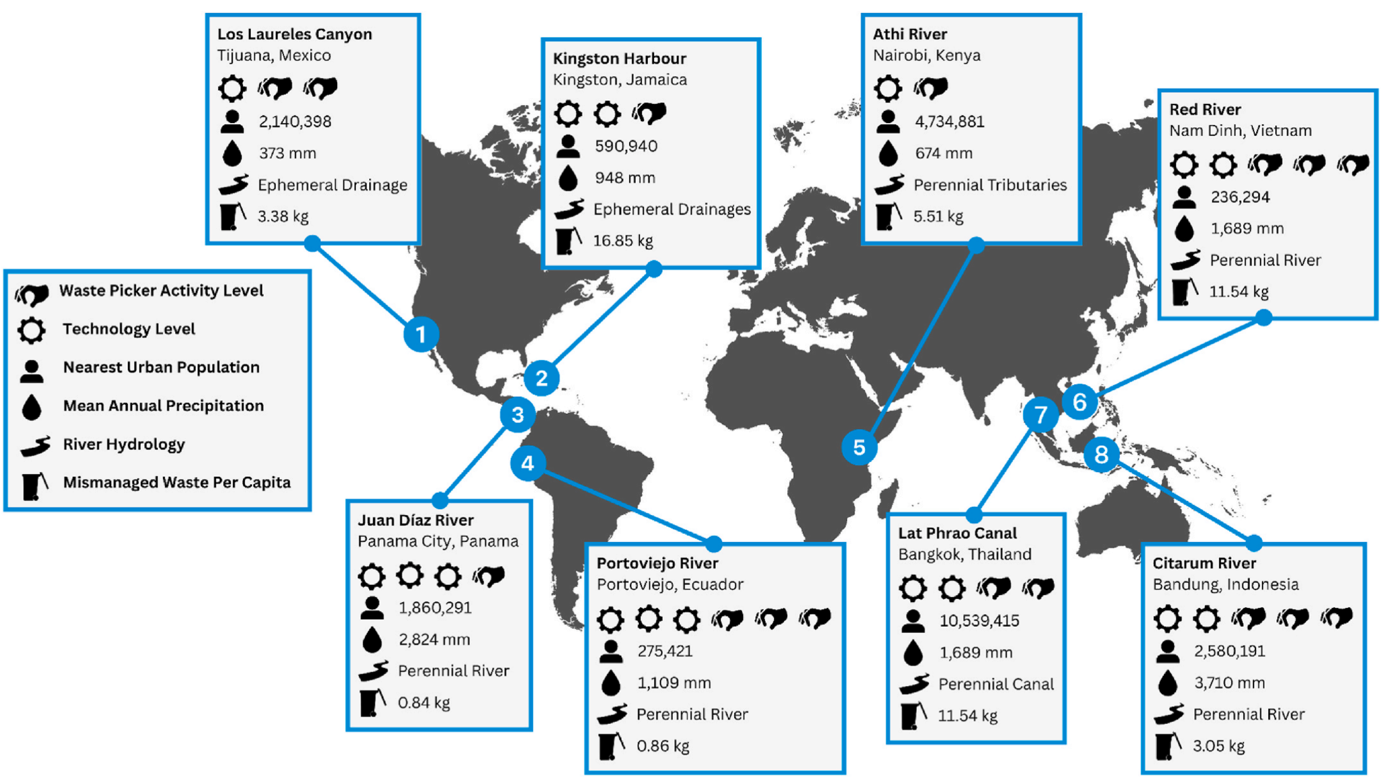

Fig. 1. The eight study sites where riverine macroplastic debris and associated data were collected. Each site is coded by icons representing the categorical level of community waste picker activity (hand) and complexity of the technology employed to collect plastic debris (gear). Low, medium, and high levels of waste picker activity and collection technology complexity are indicated by the number of icons (1–3) following the local characteristics summarized and defined in Table S1 and S2. The nearest urban population, mean annual precipitation of the nearest city, description of general hydrological features, and national mismanaged waste per capita in each respective country are provided as additional local context for each study site (see Table S1 for data sources). Photographic examples of collection technologies and debris compositions are provided in Fig. S1 and Fig. S3. Study site numbers (1–8) are used consistently as site identifiers in Figs. 1–5 in addition to study site names.

Table 1
Study Site Details. For each study site, a description of the locations where debris was collected, including location ID, debris collection technique, decimal degree coordinates, study period start and end date, and the total number of months of data collected.

Study Site	Location ID	Debris Collection Technique	Latitude (Decimal Degrees)	Longitude (Decimal Degrees)	Data Start	Data End	Total Months
Los Laureles Canyon MEX	1A	Boom serviced manually from shore;Near-river, land-based c ommunity waste collection points	32.523149	−117.091611	February 2021	December 2023	35
Kingston Harbour JAM	2A	Booms serviced primarily via boat with conveyor system	17.963069	−76.784139	March 2022	December 2023	22
Kingston Harbour JAM	2B		17.964461	−76.775381	April 2022	December 2023	21
Kingston Harbour JAM	2C		17.965431	−76.801744	May 2022	December 2023	20
Kingston Harbour JAM	2D		17.969389	−76.8095	January 2023	December 2023	12
Kingston Harbour JAM	2E		17.967923	−76.755179	July 2023	December 2023	6
Kingston Harbour JAM	2F		17.965812	−76.75905	November 2023	December 2023	2
Kingston Harbour JAM	2G		17.97348	−76.814625	November 2023	December 2023	2
Juan Díaz River PAN	3A	Highly engineered, semi-autonomous, renewable energy powered trash wheel with boom & conveyor system	9.029929	−79.442831	September 2022	December 2023	15
Portoviejo River ECU	4A	Highly engineered, semi-autonomous system with boom & conveyor system	−1.023135	−80.493174	January 2021	December 2023	36
Portoviejo River ECU	4B	Boom serviced manually from shore	−0.816442	−80.511017	December 2022	December 2023	13
Athi River KEN	5A	Variety of simple booms & barriers, serviced primarily manually from shore with some limited mechanical support; Near-river, land-based c	1.2406	36.8804	June 2020	December 2023	43
Athi River KEN	5B	ommunity waste collection points	1.306111	36.889722	July 2020	December 2023	42
Athi River KEN	5C		1.249707	36.883521	December 2020	December 2023	37
Athi River KEN	5D		−1.2442	36.891	January 2021	December 2023	36
Athi River KEN	5E		−1.2533	36.8907	January 2021	December 2023	36
Athi River KEN	5F		1.1436	36.5347	August 2021	December 2023	29
Athi River KEN	5G		1.1857	36.5114	October 2021	December 2023	27
Athi River KEN	5H		−1.243818	36.956303	November 2021	December 2023	26
Athi River KEN	5I		1.1509	36.5227	November 2021	December 2023	26
Athi River KEN	5J		1.1933	37.1161	June 2022	December 2023	19
Red River VNM	6A	Bamboo & metal trash traps with attached guiding booms serviced manually from shore	20.432111	106.189465	February 2021	December 2023	35
Red River VNM	6B		20.40424	106.25397	July 2022	December 2023	18

(continued on next page)

debris was then sorted out of this debris, dried at study sites where operationally feasible (Kenya and Thailand), and weighed. Plastic debris was further sorted by polymer classes and single-use plastic item categories and weighed as described below. Data on total debris (organic, plastic, inorganic non-plastic) and plastic debris from all eight study sites were recorded in a centralized database on a monthly time scale, reviewed and verified independently for quality and accuracy, and collated into a single, global dataset (Table S3). Because the data collection process was led by local experts representing their own communities in a diverse range of geographic, hydrologic, cultural, and socioeconomic contexts, there were slight variations in the methods utilized to amass these data between locations. Detailed descriptions of the data collection methods for each study site (S1) and how measurements were standardized across the study sites (Table S3) are provided in the supplementary materials.

Sorting macroplastic debris into seven polymer classes based on the American Society for Testing Materials (ASTM) International Resin Identification Coding System (RIC) (e.g., PET) was performed at or near the river collection study sites by local workers trained by experienced waste managers using visual inspection of ASTM indicators on plastic debris pieces as well as researched and existing local knowledge of polymer classes of brands and items. Given the scope, resources, and context of the localized data collection, laboratory analyses were not feasible for chemical confirmation of polymer samples. Two processes were used at different study sites to sort collected plastics into polymer classes using the sorting techniques outlined above: 1) in six study sites, all collected plastic debris was sorted by polymer class, and 2) in two study sites, recyclable plastic debris was sorted by polymer class and then random sub-samples of the remaining plastic debris was sorted by polymer class, and these polymer proportions were extrapolated to the total weight of the non-recyclable plastic debris (S3.2).

In four of the eight study sites, technical capacity allowed for further sorting of collected macroplastic debris into single-use plastic item categories. Four common single-use plastic item categories were recognized in these product-level characterizations: hard plastic beverage bottles (beverage bottles), film plastic food wrappers (food wrappers), flexible plastic bags (grocery and trash bags), and foam plastic packaging and food storage materials (food and packaging foam) (Table 2, S3.3).

At all study sites, the end-of-life fate of the collected macroplastic debris was recorded. The myriad of reported end-of-life fates were consolidated into five representative categories: recycled, downcycled, reused, waste to energy, and landfilled/incinerated (Hopewell et al., 2009)(S3.4). See Table 2 for the definitions and descriptions of these categories. See Table S3 for measurement standardization methods across the study sites for each plastic debris category.

2.2. *Riverine macroplastic debris data analysis, error, & statistics*

Macroplastic debris data standardization across study sites, analyses, error calculations, statistics, and visualizations were conducted using R statistical software.

Variance and standard deviation were calculated for monthly total debris and macroplastic debris across the study period for each study site. Standard error was calculated using the number of months in the study period reported respective to each study site. 95 % confidence intervals were calculated using a quartile function (Fig. 2, S3.1). Wilson score intervals were used to calculate error for the non-parametric proportion data (Figs. 3–5, S3.2-S3.4).

ANOVAs were performed to test the mean total debris and mean plastic debris collected across the study sites (Fig. 2, S3.1). Generalized linear mixed effect models (GLMM) fit via REML or maximum likelihood and t-tests using Satterthwaite's method were performed to test the influence of covariates (size of the nearest urban population, complexity of removal technology, waste picker status, river width, and river length) on both mean total debris and plastic debris collected (Fig. 2, S3.1). Random effects that account for sampling month and study site were included in all model structures. Three iterations were conducted: total debris, plastic debris, and total debris:plastic debris ratio. River length and river width were explored as potential environmental variables but were found to be highly collinear (>88 %). Therefore, the three iterations aforementioned were duplicated: each containing river width, and

Table 1 (*continued*)

Study Site	Location ID	Debris Collection Technique	Latitude (Decimal Degrees)	Longitude (Decimal Degrees)	Data Start	Data End	Total Months
Red River VNM	6C		20.262623	106.12534	November 2022	December 2023	14
Red River VNM	6D		20.3960658	106.282227	December 2022	December 2023	13
Red River VNM	6E		20.3442293	106.370038	April 2023	December 2023	9
Red River VNM	6F		20.312473	106.5216	May 2023	December 2023	8
Lat Phrao Canal THA	7A	Metal trash traps with attached guiding booms serviced *via* boat	13.779778	100.593222	August 2020	December 2023	41
Lat Phrao Canal THA	7B		13.818602	100.589287	August 2020	December 2023	41
Lat Phrao Canal THA	7C		13.8090834	100.588901	August 2020	December 2023	41
Lat Phrao Canal THA	7D		13.779883	100.5931	March 2021	December 2023	34
Citarum River IND	8A	Boom & conveyor system primarily serviced manually from shore with some limited mechanical support	−6.9180222	107.475883	December 2021	December 2023	25
Citarum River IND	8B	Manual in-river collection *via* boat	−6.9300194	107.494744	May 2023	December 2023	8

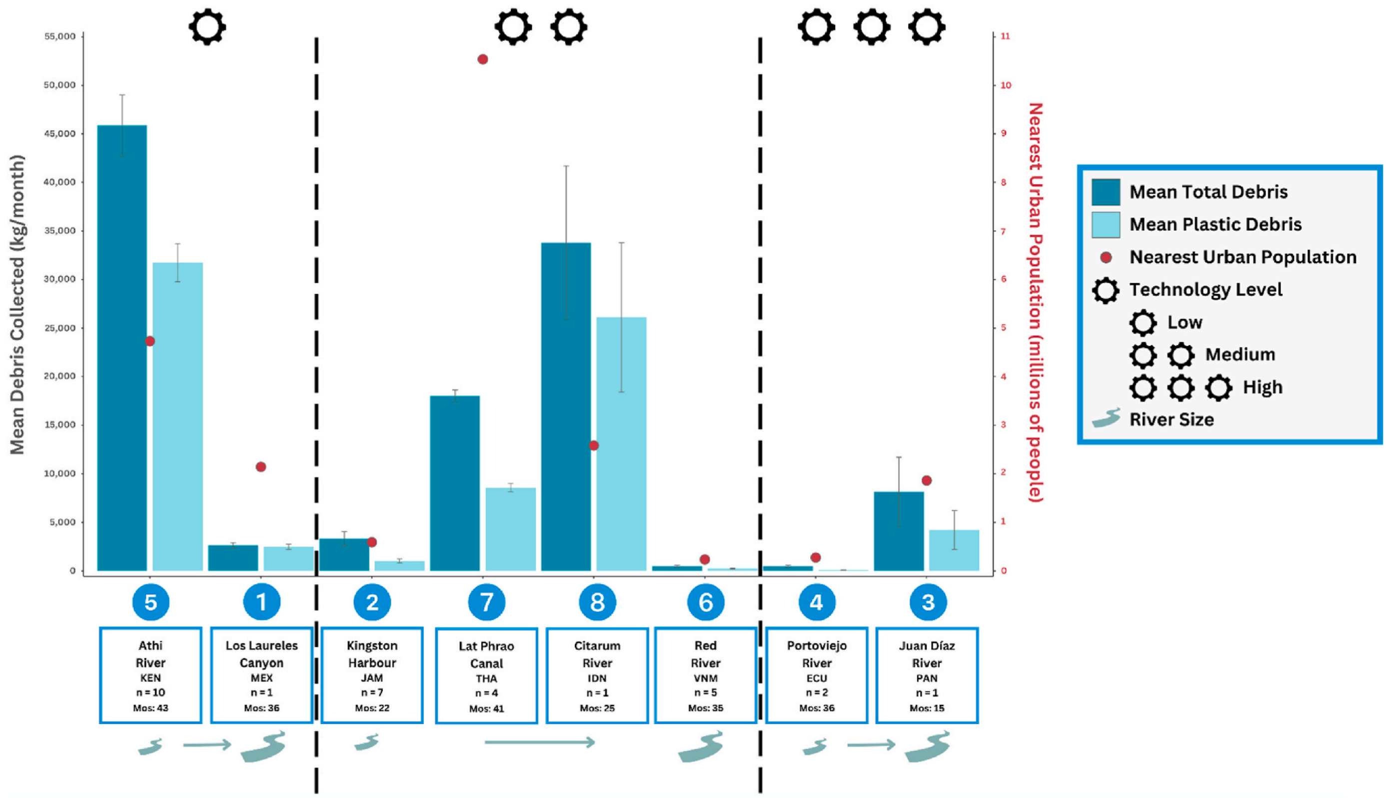

Fig. 2. The mean total debris (kg/month) collected at each of the eight study sites, and as a subset of that debris, mean plastic debris (kg/month) collected at each study site over the course of the study period (2020–2023). Study sites are separated into three categories of collection technology complexity (low, medium, high), ordered from lowest (left) to highest (right) and demarcated by vertical dashed lines. Explanation and justification of collection technology complexity categories are available in the supplementary materials (S2.1, Table S2). Within each category, study sites are sub-ordered by increasing river size from smaller (left) to larger (right), based on average river width at the specific study sites (Fig. 2, Table S1). Error bars represent the 95 % confidence intervals (Table S5). Red dots denote the population size of the nearest urban settlement to each study site (Fig. 2 and Table S1). The number (n) of collection locations within a study site and the total months of collection at each site are provided on the x-axis (Fig. 2, Table 1, Table S1). Detailed river characteristics are available in the supplementary materials (Table S1).

each containing river length.

Pearson's Chi-Square tests were performed on contingency tables of the proportions of polymers, items, and fates across all sites under the null hypothesis that polymer proportions are equal across all sites, item proportions are equal across all sites, and fate proportions are equal across all sites (Figs. 3–5, S3.2-S3.4). The nonparametric statistics approach was opted for considering these data are temporally auto-correlated, represent disparate sampling methods and efforts, and have a wide range of sample sizes, thus ruling out more robust multivariate statistical analyses for which these data do not meet the appropriate assumptions.

Detailed data management, statistics, and error methods and results are available in the supplementary materials (S3). All data and code are openly accessible *via* Zenodo with details in the supplementary materials (S4).

3. Results

3.1. Total debris & plastic debris collected

In total, we collected and analyzed 3,842,576 kg of debris (including organic matter, metals, glass, plastic, and other materials) across the eight study sites during this study period, collectively representing 250 months of sampling effort. Of that total debris, 2,534,260 kg (66 %) was plastic debris.

Mean total debris collected across all study sites was 15,370 kg/month. Mean total debris collected at the individual study sites ranged from a low of 484 kg/month in the Portoviejo River, Ecuador to a high of 45,863 kg/month in the Athi River, Kenya (Fig. 2, Table S5). Of the total debris, mean plastic debris collected across all study sites was 10,137 kg/month. Mean plastic debris collected at the individual study sites ranged from a low of 93 kg/month in the Portoviejo River, Ecuador to a high of 31,731 kg/month in the Athi River, Kenya (Fig. 2., Table S5). While we saw significant differences in total debris ($F_{7,\ 52.98}$, $p < 0.001$) and plastic debris ($F_{7,\ 51.48}$, $p < 0.001$) collected across our eight study sites, potential explanatory variables such as the categorical level of collection technology complexity, categorical level of community waste picker activity, average river width at the specific study sites, river length, and population size of the nearest urban settlement to each study site did not statistically affect the total debris ($p > 0.05$) or plastic debris ($p > 0.05$) collected (S3.1). Detailed results, statistical methods, and error are available in the supplementary materials.

3.2. Polymer composition of plastic debris collected

We analyzed the polymer composition of the plastic debris collected based on the American Society for Testing and Materials (ASTM) International Resin Identification Coding System (RIC) codes 1–7 (Table 2), which revealed a high degree of variability among the eight study sites (Fig. 3, Table S6). Overall, the most common polymer class across all study sites was low-density polyethylene (LDPE), with the site-specific composition of LDPE varying from a low of 1.2 % (Juan Díaz River, Panama) to a high of 65.4 % (Lat Phrao Canal, Thailand) (Fig. 3, Table S6). Polyethylene terephthalate (PET) was the second most common polymer class and contributed the highest proportion of plastic debris collected in three of the eight study sites: Athi River, Kenya (28.5 %); Juan Díaz River, Panama (59.5 %); and Kingston Harbour, Jamaica (72.9 %) (Fig. 3, Table S6). In Los Laureles Canyon, Mexico, PET

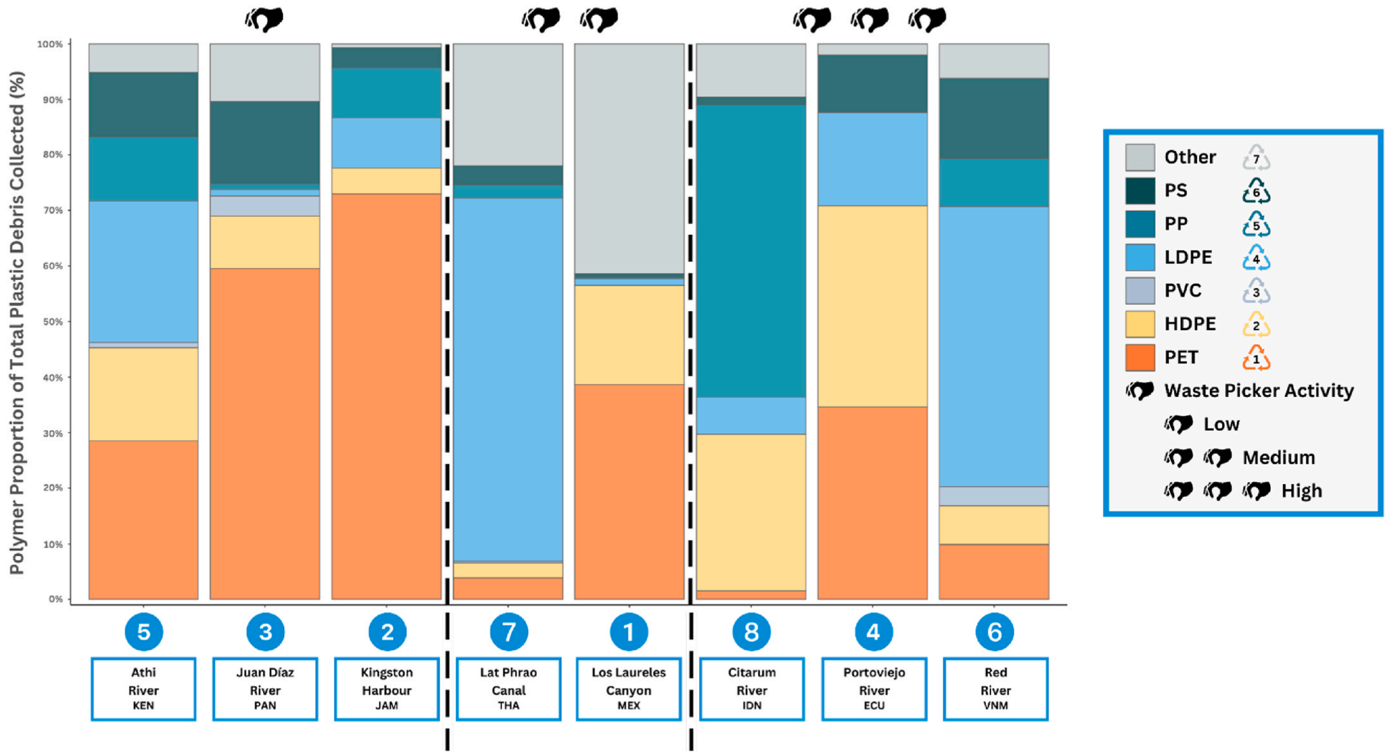

Fig. 3. The polymer composition of plastic debris collected at each of the eight study sites by weight (kg), presented as a percentage of total plastic debris collected at that site. Study sites are separated into three categories of community waste picker activity (low, medium, high) in the local area upstream of the collection point, ordered from lowest (left) to highest (right) activity and demarcated by vertical dashed lines. Explanation and justification of waste picker activity categories are available in the supplementary materials (S2.1, Table S2). Definitions of polymer classes (based on the American Society for Testing Materials (ASTM) International Resin Identification Coding System (RIC)) are described in Table 2.

comprised the second highest proportion (38.7 %) after waste automotive tires (41.4 %, noted as "Other") (Fig. 3, Table S6). High-density polyethylene (HDPE) comprised the highest proportion of plastic debris collected in the Portoviejo River, Ecuador (36.2 %); LDPE comprised the highest proportion in the Red River, Vietnam (50.4 %); Polypropylene (PP) comprised the highest proportion in the Citarum River, Indonesia (52.6 %) (Fig. 3, Table S6). The proportional compositions of polymer classes across study sites were substantially different ($X^2_{42,751.39}$, $p < 0.001$) (S3.2).

3.3. Proportion of single-use plastic item categories in plastic debris collected

Data collection on common single-use plastic item categories collected (grocery and trash bags, food wrappers, beverage bottles, food and packaging foam) was carried out at four of the eight study sites (Athi River, Kenya; Citarum River, Indonesia; Portoviejo River, Ecuador; Red River, Vietnam; collection methods utilized in the other four study sites did not support enumeration of single-use plastic item categories). These single-use plastic item categories in sum accounted for 63.2 % of the total plastic debris collected across these four study sites. In the Athi River, Kenya and the Portoviejo River, Ecuador, beverage bottles were the most abundant single-use plastic item category collected (27.9 % and 21.5 %, respectively), and grocery and trash bags comprised 14.3 % and 7.9 %, respectively (Fig. 4, Table S7). In the Citarum River, Indonesia and the Red River, Vietnam, grocery and trash bags were the most abundant single-use plastic item category (23 % and 46.2 %, respectively) (Fig. 4, Table S7). Proportionally, grocery and trash bags in the Red River, Vietnam represented the largest share of any single-use plastic item category among all study sites relative to the total plastic debris collected (46.2 %) (Fig. 4, Table S7). Beverage bottles (1.5 %) and food and packaging foam (1.5 %) in the Citarum River, Indonesia represented the smallest share of any single-use plastic item category among all study sites (Fig. 4, Table S7). The proportional compositions of single-use plastic item categories across these four study sites were substantially different ($X^2_{12,124.48}$, $p < 0.001$) (S3.3).

3.4. End-of-life fate of plastic debris collected

Of the total plastic debris collected across all study sites, 14 % was recycled, 62.9 % downcycled, 3 % reused, 12.3 % processed as waste to energy, and 7.8 % landfilled (see Table 2 for definitions of end-of-life fates). The proportional compositions of end-of-life fate varied substantially across study sites ($X^2_{28,1618.97}$, $p < 0.001$) (S3.4). Some proportion of plastic debris collected at all study sites was recycled (except Athi River, Kenya) (Fig. 5, Table S8). The proportion of plastic debris recycled at the individual study sites ranged from a low of 10.7 % (Lat Phrao Canal, Thailand) to a high of 65.1 % (Portoviejo River, Ecuador) (Fig. 5, Table S8). Plastic debris collected was also landfilled at all study sites (except Lat Phrao Canal, Thailand), with proportions ranging from a low of 6 % (Citarum River, Indonesia) to a high of 73.5 % (Red River, Vietnam) (Fig. 5, Table S8). In Thailand, the vast majority of plastic debris collected was processed as waste to energy (89.3 %); no other study site utilized waste to energy (Fig. 5, Table S8). Plastic debris collected was reused at two study sites with proportions ranging from a low of 3.3 % (Athi River, Kenya) to a high of 34.9 % (Los Laureles Canyon, Mexico) (Fig. 5, Table S8). A large proportion of plastic debris collected was downcycled in Kenya (89.8 %) and Indonesia (58.7 %), the only study sites where this method was utilized (Fig. 5, Table S8).

4. Discussion

Rivers represent an important focal system for characterizing plastic debris that contributes to plastic pollution, not only to understand rates

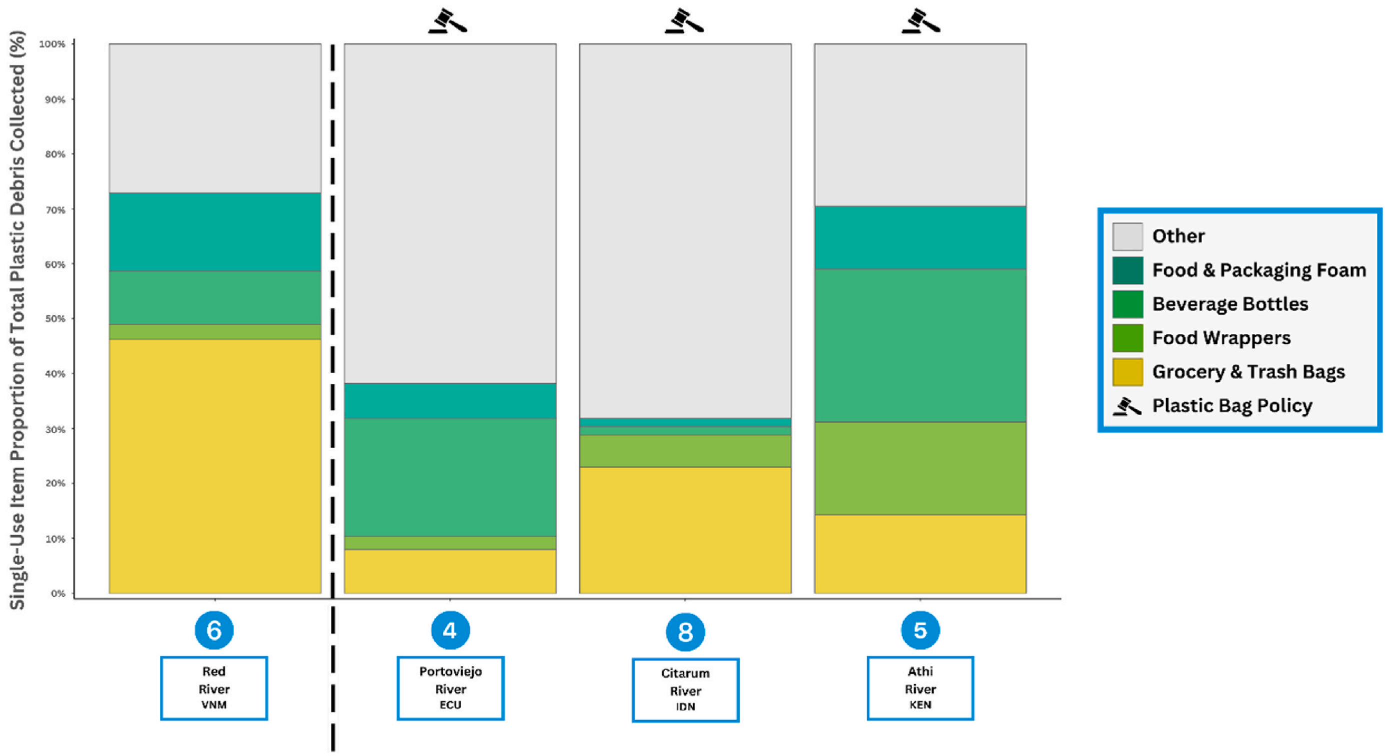

Fig. 4. The proportion of single-use plastic item categories collected at four of the eight study sites by weight (kg), presented as a percentage of the total plastic debris collected within that site. "Other" includes all other plastic collected and weighed at each study site that did not fall into the specific single-use plastic item categories presented. Three of the four study sites were located in or near jurisdictions that contain some level of plastic bag ban or control policy (Athi River, Kenya; Citarum River, Indonesia; and Portoviejo River, Ecuador; denoted with gavel symbol). These three study sites with plastic bag policies are ordered from weakest (left) to strongest (right) in deference to the scope of these policies and the level of enforcement and are separated from the study site with no pertinent plastic bag policy by a vertical dashed line (Red River, Vietnam). Further detail on relevant policies and laws is provided in Table S4.

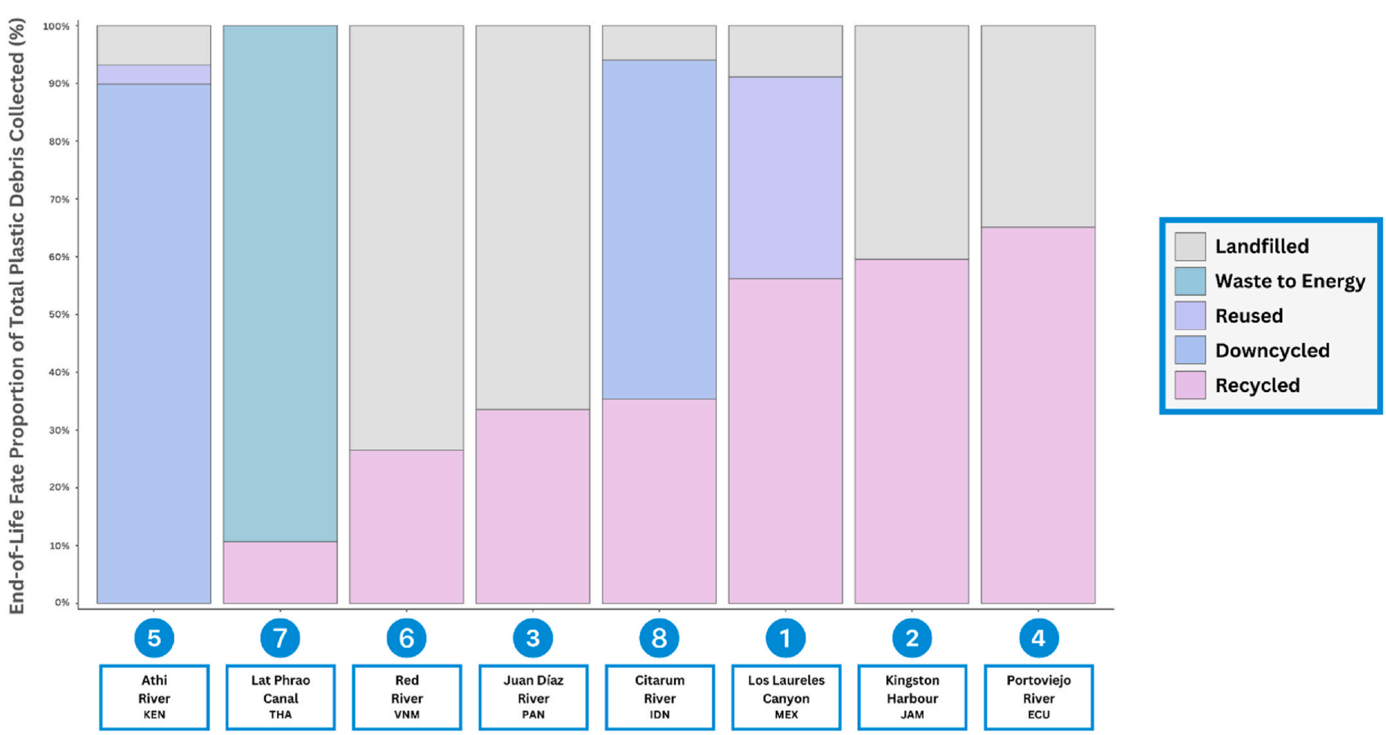

Fig. 5. The end-of-life fate of the plastic debris collected within each of the eight study sites by weight (kg), presented as a percentage of the total plastic debris collected within that study site over the course of the study period (2020–2023). The study sites are presented in ascending order of the proportion of collected plastic that was recycled, from lowest (left) to highest (right). End-of-life fates are defined in Table 2.

Table 2
Data Metrics Definitions and Descriptions Plastic polymers are indicated by their resin identification code based on the American Society for Testing Materials (ASTM) International Resin Identification Coding System (RIC), along with their full chemical name and common items made from each polymer. It is important to note that the example items are not exhaustive or exclusive to each polymer. The same item can be made with different polymers, and often, polymers are combined to manufacture these items. In addition, even within a polymer category, there are a range of chemical additives in various items, so two items both made from the same polymer can be vastly different depending on the additional chemicals used in the manufacturing process.

Metric	Category	Description
Total Debris & Plastic Debris	Total Debris	All debris collected (organics, plastics, metals, glass, other materials)
	Plastic Debris	Plastic debris collected, as a subset of total debris collected
Plastic Polymer Classes	PET	Plastic #1: Polyethylene Terephthalate (e.g., beverage bottles) [hard plastic]
	HDPE	Plastic #2: High-Density Polyethylene (e.g., laundry detergent containers, toys) [hard plastic]
	PVC	Plastic #3: Polyvinyl Chloride (e.g., pipes and construction materials) [hard plastic]
	LDPE	Plastic #4: Low-Density Polyethylene (e.g., grocery bags, food wrappers) [film plastic]
	PP	Plastic #5: Polypropylene (e.g., food storage containers, woven rice bags) [hard and film plastic]
	PS	Plastic #6: Polystyrene (e.g., foam packaging) [film and foam plastic]
	Other	Plastic #7: All Other Plastic (e.g., rubber; sunglasses, tires)
Single-Use Plastic Item Categories	Grocery & Trash Bags	Film, flexible single-use bags; primarily LDPE and sometimes HDPE
	Food Wrapper	Film, flexible single-use film food wrappers; primarily LDPE and PP
	Beverage Bottles	Hard single-use beverage bottles; primarily PET
	Food & Packaging Foam	Single-use foam packaging and food storage materials; primarily PS
End-of-Life Fates (Moffett, 2024)	Recycled	Plastic sent to traditional “primary” or “mechanical” recycling facilities, directly or indirectly through a third-party consolidator, for reprocessing into a product with equivalent properties
	Downcycled	Plastic directly used, stockpiled, or sold to a third-party, for the purpose of reprocessing into new materials with lesser properties and value (e.g., plastic paving brick, plastic board)
	Reused	Plastic used directly for new, alternative, or similar purposes that does not involve processing the plastic (e.g., using tires as planter boxes)
	Waste to Energy	Plastic used in a formal energy recovery process (e.g., gasification through pyrolysis, electricity cogeneration)
	Landfilled	Plastic sent to formal, controlled, and sanitary landfill or incineration facilities, with no energy recovery process used in the case of incineration

and drivers of plastic emissions into and out of rivers, but also to shed light on the root causes by which this plastic pollution is generated in the first place and how to better manage it. We examined how and why plastic debris amounts (Fig. 2), compositions (Figs. 3 and 4), and end-of-life fates (Fig. 5) differed between and within our study sites and what those patterns reflect about local plastic production, consumption, and waste management, as well as their implications for tactically reducing upstream entry of plastic debris into the environment in the first place.

4.1. Total debris & plastic debris collected (Fig. 2)

Collection technology complexity varied between our eight study sites, ranging from teams of local waste pickers employed for manual removal of plastic debris trapped by simple booms placed in rivers, river banks, and areas adjacent to these rivers (e.g., Los Laureles Canyon, Mexico and Athi River, Kenya) to highly-engineered solar- and water-powered boom and conveyor systems (e.g., Juan Díaz River, Panama and Portoviejo River, Ecuador) that semi-autonomously collected and removed plastic debris from rivers (Table S1, Fig. S1). While other important advantages are conferred by the use of the more advanced – and thus more expensive – plastic collection technologies (e.g., capturing media interest and public dialogue about plastic pollution in addition to capturing plastic debris), these characteristics do not alone appear to be prerequisite for achieving high yields of plastic debris collection in rivers.

Rather than technological complexity of riverine plastic collection methods, inter-site variation in operational approaches at our study sites appeared to be more consequential to the collection rates observed in these data. For example, in the Red River, Vietnam, 15 m-wide bamboo and metal trash traps were installed along the riverbank, yet the river itself spans 800 m at some points (Table S1). The dispersion of plastic debris across an especially large river relative to the small size of the traps may have limited their collection efficacy and contributed to the observed lower collection rates at this study site. By comparison, trash traps of a similar size and complexity were used in the 20 m-wide Lat Phrao Canal, Thailand (Table S1). Placed in a series to span the entire waterway, collection rates were higher here than those observed in the Red River, Vietnam. Collectively, observations like these strongly support the notion that there is no “one size fits all” solution for maximizing plastic debris collection in rivers and that instead collection efforts perform best when they are strategically tailored to fit the local context (Schmaltz et al., 2020; Helinski et al., 2021; Falk-Andersson et al., 2020).

Patterns in the collection rates also appeared to be influenced more strongly by the social context (population size) than by the environmental conditions (river width and length) or the collection technology complexity (Table S1, S3.1). The study sites in Kenya, Indonesia, and Thailand had the highest plastic debris collection rates, respectively, corresponding to the largest population sizes of the nearest urban settlements: Bangkok, Thailand (10,539,415 people); Nairobi, Kenya (4,734,881 people); and Bandung, Indonesia (2,580,191 people) (Figs. 1 and 2, Table S1). Vietnam and Ecuador – the study sites with the lowest plastic debris collection rates – corresponded to the smallest population sizes of the nearest urban settlements: Nam Dinh, Vietnam (193,499 people) and Portoviejo, Ecuador (275,421 people) (Figs. 1 and 2, Table S1). This observation mirrored observations in other studies suggesting that proximity to highly populated cities was a strong determinant of riverine plastic emissions and other adjacent environmental contexts (Meijer et al., 2021; Jambeck et al., 2015). This is informative when considering where to site future collection efforts so as to maximally reduce plastic debris emissions.

While seasonality has been shown in the literature to be a significant factor influencing debris flows in rivers (Axelsson et al., 2017; van Emmerik et al., 2019b), the influence of other practical and operational factors at the study sites may have outweighed the influence of precipitation and other temporal and interannual factors on debris collected. Examples include Portoviejo, Ecuador (where the collection technology was protected by removing it from the river during peak rain season due to heavy and damaging large organic debris mobilization such as full trees); Bangkok, Thailand (where peak operational and technological efficiency allowed for consistent collection rates year round); Tijuana, Mexico (where community-based near-river land collection supplemented collection rates within the ephemeral drainage during dry seasons); and Bandung, Indonesia (where rainy season coincided with increased presence of water hyacinths that made debris collection significantly more difficult). This provides an opportunity for further exploration in which the study design could control for these potentially confounding factors. Time series visualizations of monthly total debris (kg) and plastic debris (kg) for each study site during their respective study periods are available in the supplemental materials (Fig. S2).

4.2. Polymer composition of plastic debris collected (Fig. 3)

The observed substantial differences across study sites in polymer composition provide some insight into variance in plastic production, consumption, and waste management practices and challenges between these regions. It is apparent that no one factor drives the observed variation across all study sites, but rather is a result of multiple intersecting influences. Collection methods, local and national policies, waste management access and infrastructure, and consumer behavior are all factors that may have contributed to polymer composition variations. Empirical data paired with local cultural and institutional knowledge, however, is helpful to further examine the impact of these individual factors in different contexts to better target efficient upstream solutions.

The high prevalence of more readily recyclable polymer classes, such as PET, in the plastic debris collected in this study reflects the dearth of recycling infrastructure (both collection and processing structures) in these communities and simultaneously an economic opportunity for future growth and investment in such infrastructure to take advantage of this supply of candidate feedstock (Fig. 3). High proportions of PET, for example, were recorded in the Juan Díaz River, Panama and Los Laureles Canyon, Mexico (59.5 % and 38.7 % respectively), suggesting there is strategic value in the recent investments in the recycling industry in these countries and regions ($20 Million recycling facility in Panama - CentralAmericaData :: the regional business portal; Alpla, 2022; Coke bottlers invest in PET, 2023; Recycling in Central America - CentralAmericaData; Recycling in Panama - CentralAmericaData) (Fig. 3, Table S6). However, we measured the highest proportions and high overall volumes of PET in Kingston Harbour, Jamaica (73 %) where there has to date been less recycling investment, highlighting the added challenges faced by island nations to manage plastic pollution (Fig. 3, Table S6).

Conversely, we observed considerably lower proportions of PET in the plastic debris captured in study sites such as Thailand, Indonesia, and Vietnam, where there was a strong presence of informal waste picking activity (Fig. 3, Table S2, Table S6). PET is one of the most readily recycled and thus often one of the highest value polymer classes to waste pickers (Hossain et al., 2024; Suhaimi et al., 2022), which suggests that this informal sector may have collected a large share of the PET in these river systems before it reached the study sites (Fig. 3, Table S6). Therefore, the reduced proportions of PET at these study sites does not necessarily indicate that these regions are producing and consuming less PET but rather that there is likely more circularity in these regions. The patterns in these data underline the critical role that the informal waste picker sector can play in plastic pollution management and the effective promotion of plastic recycling and circularity (Dias, 2016; Chen et al., 2018).

The largest fraction of plastic debris categorized as the "Other" polymer class was reported in Los Laureles Canyon, Mexico, of which the majority was waste automotive tires (Fig. 3, Fig. S4, Table S6). While variable in construction, about 60 % of tire rubber is composed of synthetic plastic polymers (Eranki and Landis, 2019; Ramarad et al., 2015). While tires were detected and collected at other study sites, none revealed a prevalence approaching those recorded at the Mexico site (Fig. 3, Table S6). Waste tire pollution in the Mexico-USA border region is a well-documented problem, stemming from the formal and informal import of used tires of variable quality from US border states into Mexico (Spitz; The Flow of Used and Waste Tires, 2009; The Flow of Used Tires, 2017; Border 2012 Accomplishments Report, 2014). The patterns in these data underline the urgency of adopting and extending national and international programs and actions such as extended producer responsibility, taxes, or fees (Campbell-Johnston et al., 2020; Winternitz et al., 2019) to curtail this unique form of plastic waste mismanagement. Such interventions would benefit countries, such as Mexico, hosting and burdened from this waste import, but also transboundary marine and coastal ecosystems and economies into which much of this waste may ultimately be vectored – systems that are shared by countries driving these exports.

4.3. Proportion of single-use plastic item categories in plastic debris collected (Fig. 4)

The single-use plastic item category data collected at different study sites afforded the opportunity to examine relationships between these data and local policies restricting or banning single-use plastic items. For example, we considered bans on single-use plastic bags. Across the four study sites where single-use plastic item category data was collected, there existed a spectrum of single-use plastic bag ban policies (Table S3). Some form of policy aimed at control of single-use plastic bags had been instituted in all study sites except Nam Dinh, Vietnam. Though more targeted research aimed at evaluating the impacts of differences in policy scopes, enforcement capabilities, and shifts in waste streams is required to truly understand the effectiveness of specific policies, we do provisionally note that the proportion of single-use plastic bags was considerably lower (ranging from a low of 7.9 % to a high of 23 %) in our data from study sites where plastic bag ban policies do exist, versus 46.2 % in Vietnam where such policies are lacking (Fig. 4, Table S7). The planned implementation of a single-use plastic bag ban in Vietnam in 2026 (Policy Brief, 2022) will provide an interesting natural experimental opportunity to conduct before/after sampling to more clearly resolve the mechanistic role of such policies in shaping these patterns. Research conducted in other areas before and after the implementation of similar kinds of single-use plastic bans (including research in rivers relating to single-use plastic bag bans) add support to the view that indeed such policies can and do positively result in reductions of regulated plastics in these environmental waste streams (Bag Law Survey Overview, 2013; Adeyanju et al., 2021; Convery et al., 2007).

The composition of plastic debris has been observed to vary between the different environmental contexts through which plastic debris travels and is sampled (Papp and Oremus, 2025). For example, the composition of plastic debris measured in coastal marine ecosystems (e.g., considerable apparent contributions from single-use and packaging plastic) appears to differ from the composition of plastic measured in open ocean gyres (e.g., large contribution of plastic debris originating from the fishing industry) (Lebreton et al., 2022; Choy et al., 2019). To identify potential environmental context differences in the case of this new riverine plastic debris dataset, we compared the observed frequencies of single-use plastic items in our river study sites to their prevalence in beach and coastal plastic debris collection and research efforts conducted synchronously in the same four countries (S2.3) (Ocean Conservancy, 2024). The key difference between these environmental contexts was the low proportion of grocery and trash bags in beach and coastal contexts (6 %) compared to river contexts (16.1 %) (Fig. S5, Table S9). These findings are consistent with previous studies that suggest that film plastic such as plastic bags might have higher retention rates in rivers and slower transport progress due to entanglement in vegetation and/or accumulation with organic matter, which in turn may contribute to a higher proportion of these items in rivers versus coasts (Ivar do Sul et al., 2014; Schwarz et al., 2019). These comparisons reveal geographic and environmental patterns of plastic consumption, waste generation, and transport and may provide insights for more efficient management and prevention of plastic pollution tailored to specific contexts.

4.4. End-of-life fate of plastic debris collected (Fig. 5)

Further context around the end-of-life fate of plastic debris collected in this study illuminates significant on-the-ground challenges to the responsible disposal of debris. For example, to achieve the high recycled rate (56.2 %) at the study site in Los Laureles Canyon, Mexico, this mass of plastic debris was driven by truck nearly 2700 km from Tijuana to a recycling plant in Toluca because no plastic recycling infrastructure exists in the region (Fig. 5, Table S8). Similarly, in Kingston Harbour, Jamaica, the 59.5 % recycled rate is caveated with the reality that this plastic debris was stockpiled by a third party consolidator on the island and eventually shipped overseas as far as eastern Europe (Fig. 5, Table S8). These inefficient distances are associated with high transport costs, high carbon emissions, and a general loss of transparency in the supply chain as to the final true amount of material that is ultimately transformed into useable recycled feedstock.

Downcycling, or the processing of collected plastic debris as feedstock for the creation of a new but less valuable product, was observed at the study sites in Kenya and Indonesia, where 89.8 % and 58.6 % of the collected plastic debris was downcycled respectively into construction materials to be sold (Fig. 5, Table S8). While local technical capacity and infrastructure to recycle some plastic debris does exist in Kenya, the low local value of recycled materials made it more economically viable to use it as feedstock for this downcycling business. Similarly, in Indonesia, plastic boards have been produced from low-value, locally non-recyclable flexible plastic debris.

Lat Phrao Canal, Thailand was the only study site that routed a substantial proportion (89.3 %) of its collected plastic debris to a controlled and vetted waste to energy plant (Fig. 5, Table S8). This resulted from the study site operator's "no waste-to-landfill" policy, compared to a lack of access to waste to energy processing facilities and/or prohibitive costs that prevented this option in the other study sites. It is notable that the local team in Thailand invested extensive effort and leveraged considerable technical expertise in matching collected plastic debris to recycling opportunities and yet because of the lack of infrastructure, still was only able to recycle 10.7 % of collected plastic debris (Fig. 5, Table S8).

Together these observations reflect the complexity of waste management in real practice, and while out-of-scope of this study, point towards the need for more research into the economic and infrastructural viability of scaling the many waste management solutions potentially available.

4.5. Study limitations

There are a number of important caveats to consider when interpreting the communicated patterns. First, the methods employed in this study primarily provide insight into buoyant macroplastic debris found within and immediately adjacent to these rivers. These findings do not necessarily provide direct insight into all plastic debris that might be transported in the lower water column or that is deposited under riverbed sediments – depending on the river characteristics and the technology employed (i.e., these data do not reflect or account for 100 % of plastic debris transported by these rivers) (Wang et al., 2024; Liro et al., 2020; Newbould et al., 2021). This study also does not measure the important role of micro- and nanoplastic emissions, which are known to be high in rivers and that can present some of the same or even more challenging threats to human and environmental health (Margenat et al., 2021; Kumar et al., 2021; Windsor et al., 2019). Regional differences in polymer class labeling and associated definitions (e.g., the same woven plastic rice bag may be labelled HDPE in one region and PP in another, especially where there is minimal oversight, regulation, and consumer protection) may have similarly influenced data reporting in addition to possible biases due to differences in categorization and reporting of end-of-life fates. Lastly, this study considers data from eight river systems that may not fully represent the diversity of regional, national, and global river systems, limiting the applicability of findings to other regions. These limitations underscore the need for further data collection and research related to the generation, composition, impact, and management of riverine plastic debris.

4.6. Conclusions

We submit the following four observations regarding the potential actions that may help reduce emissions of plastic debris in rivers and other aquatic and marine ecosystems.

1. Create value for collected plastics

When certain plastic polymer classes and items held some local value, they were less likely to be found in rivers. The low prevalence of PET beverage bottles in Bandung, Indonesia, where waste pickers already collected this PET, provides one compelling line of support for this observation (Fig. 3, Table S2, Table S6). Institutionalized methods utilized to give plastic waste value such policies for minimum recycled content and bottle deposit fees would appear to contribute to reducing such forms of plastic waste mismanagement (Pottinger, 2024; SB 54; OECD, 2024; Kutkaitis et al., 2024).

2. Spur investment in waste management and recycling infrastructure and services

With the high plastic debris emissions rates measured in this study (Fig. 2), considerable levels of leakage are clearly occurring in local waste management systems, if such basic services are present at all. Enhanced investment in waste management, especially in geographies where investments are historically lacking, are essential for reducing this mismanagement.

Our end-of-life fate data also illustrated that robust and accessible recycling infrastructure and services were lacking in the regional vicinity of virtually all of the study sites, as evidenced by the low overall recycled rate observed in this study (14 %) (Fig. 5, Table S8). Even in cases where recycling infrastructure was present in-country, it was often expensive, inconvenient (requiring trucking over long distances), or uncertain (if exported, the ultimate fate was in question). Promoting the political and economic conditions that enable local, accessible, and affordable recycling would benefit these and other analogous communities. Investments in the informal economy (e.g., waste pickers) also confer a valuable opportunity to strengthen the recycling chain while creating employment. A just transition must be made where these actors are not left behind but are included in efforts to increase recycling rates (Velis, 2017; Moffett, 2024).

3. Enhance data collection

Data collection is critical to furthering our understanding of the necessary upstream (e.g., production and consumption patterns) and downstream (e.g., waste management) solutions to curb plastic pollution. Regular data collection should be coupled with formal and informal cleanup programs that track plastic emissions through time and are a necessary piece of the multi-pronged actions required to address plastic pollution. Data that identify debris dynamics and hotspots can improve the success of future cleanup efforts themselves. Efforts to create standardized data protocols for independent groups using any type of debris collection technology to submit data on plastic pollution increases the collective global impact potential of such efforts (Ocean Conservancy, 2024; Sherlock et al., 2023; Jambeck and Johnsen, 2015). Areas of promising future research in this domain include monitoring efforts in rivers before and after policy implementation, and large-scale standardized efforts to measure microplastic pollution in rivers and riverine plastic transport and deposition behaviors.

4. Enact supportive local, national, and international policy

Strong local policies have a positive impact on reducing plastic debris in some of our study sites. This was evidenced in Nairobi, Kenya where a single-use plastic bag ban was associated with a lower relative proportion of plastic bags in the collected plastic debris at our study site (Fig. 4, Table S7). However, the scope and details of these bans matter. In Indonesia, for example, single-use plastic bag bans only cover supermarkets (overlooking the more popular traditional outdoor markets), and the measured proportion of single-use plastic bags at our study site would suggest this policy is consequently having a more muted impact (Fig. 4, Table S7).

There are clear limits, however, to the overall impact of local and even national policies. International policy interventions, such as the global policy instrument to end plastic pollution under negotiation by the United Nations, that account for the global nature of the trade and business of plastic production and waste appear necessary to more holistically reduce single-use plastic, create the scale of financing required to bolster waste management and recycling infrastructure, and to create market conditions that add value to end-of-life plastic.

These results and the dataset from which these conclusions are drawn provide uniquely diverse, empirical insight into patterns of riverine macroplastic debris. They also provide a perspective on challenges and opportunities for constructively handling plastic debris collected from rivers. Lastly, these data provide guidance into the impact of specific actions that have been taken or could be taken to reduce the input of plastic debris into rivers in the first place.

The observed high rates of macroplastic debris in these river systems underscores the severity of plastic pollution as an urgent global issue. This work and future analyses of these types of debris, however, also illuminate some of the most promising pathways that can be taken to begin addressing this important social and environmental challenge.

CRediT authorship contribution statement

Chase W. Brewster: Writing – original draft, Writing – review & editing, Visualization, Validation, Project administration, Methodology, Investigation, Formal analysis, Data curation, Conceptualization. **Molly R. Morse:** Writing – original draft, Project administration, Methodology, Investigation, Funding acquisition, Formal analysis, Data curation, Conceptualization. **Robert J. Fournier:** Writing – original draft, Writing – review & editing, Validation, Formal analysis. **Lucas Joseph:** Formal analysis, Data curation. **Britta R. Baechler:** Writing – original draft, Project administration, Data curation. **Hannah De Frond:** Writing – original draft, Data curation. **Thaine Herman Assumpcao:** Data curation. **A. Kita Pritasari:** Data curation. **Yunisa Zahrah:** Data curation. **Gulontam Situmorang:** Project administration. **Anssi Mikola:** Project administration. **Nicole Becerra:** Project administration, Data curation. **José Pérez:** Data curation. **Inty Grønneberg:** Project administration. **Alvaro Quiros:** Investigation, Data curation. **Mirei Endara de Heras:** Project administration. **Sandy Watemberg:** Project administration. **Clifford Okoth:** Project administration. **Martina Sikawa:** Data curation. **Moses Okoth:** Data curation. **James Scott:** Project administration, Data curation. **Ma del Rosario Norzagaray Román:** Data curation. **Fay Crevoshay-Engelmayer:** Project administration. **Angela Kemsley:** Project administration. **Vien Tran:** Project administration. **Sandra Whitehouse:** Project administration. **Ho Thi Yen Thu:** Project administration, Data curation. **Stephanie Ritchie:** Project administration. **Dominik Haertl:** Project administration. **Michael McCarthy:** Investigation, Data curation. **Caroline Mahfood:** Project administration. **Douglas J. McCauley:** Writing – original draft, Methodology, Investigation, Funding acquisition, Conceptualization.

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Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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In addition to the data insights presented in this manuscript and the positive local impacts created by this international effort, a considerable collection of practical insights, field experiences, and lessons learned were generated that we believe to be useful for future large-scale studies of this nature. We have collated and communicated this valuable information on the Clean Currents Coalition website, and we encourage others interested in this type of work to use, share, and reference these works.

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Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.jenvman.2025.126354.

Data availability

Data, metadata, and R code necessary to reproduce and build upon results, analyses, statistics, and figures are accessible *via* Zenodo. [https://zenodo.org/records/15759241]

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