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Massive plastic pollution in a mega-river of a developing country: Sediment deposition and ingestion by fish (*Prochilodus lineatus*)[☆]

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ABSTRACT

The aim of this study was to determine the amount, composition and origin of plastic debris in one of the world largest river, the Paraná River in Argentina (South America), focusing on the impact of urban rivers, relationships among macro, meso and microplastic, socio-political issues and microplastic ingestion by fish.

We recorded a huge concentration of macroplastic debris of domestic origin (up to 5.05 macroplastic items per m^2) dominated largely by bags (mainly high- and low-density polyethylene), foodwrapper (polypropylene and polystyrene), foam plastics (expanded polystyrene) and beverage bottles (polyethylene terephthalate), particularly downstream from the confluence with an urban stream. This suggests inadequate waste collection, processing and final disposal in the region, which is regrettably recurrent in many cities of the Global South and Argentina in particular.

We found an average of 4654 microplastic fragments m^{-2} in shoreline sediments of the river, ranging from 131 to 12687 microplastics m^{-2}. In contrast to other studies from industrialized countries from Europe and North America, secondary microplastics (resulting from comminution of larger particles) were more abundant than primary ones (microbeads to cosmetics or pellets to the industry). This could be explained by differences in consumer habits and industrialization level between societies and economies.

Microplastic particles (mostly fibres) were recorded in the digestive tract of 100% of the studied *Prochilodus lineatus* (commercial species).

Contrary to recently published statements by other researchers, our results suggest neither macroplastic nor mesoplastics would serve as surrogate for microplastic items in pollution surveys, suggesting the need to consider all three size categories.

The massive plastic pollution found in the Paraná River is caused by an inadequate waste management. New actions are required to properly manage waste from its inception to its final disposal.

1. Introduction

Plastic pollution is one of the great challenges for environmental management in our times. Plastic debris is a combination of high persistence, low density, and extremely wide size distribution. This causes the behavior of plastic debris to show a far wider variety than most other materials, such as suspended fine sediments (Kooi et al., 2018). Plastic particles cause severe damage to freshwater and marine ecosystems (Galloway et al., 2017). In the oceans alone, the economic damage due to plastic pollution is estimated as high as 21 billion Euro (Beaumont et al., 2019). In spite of a great scientific effort to tackle this problem worldwide the state of our knowledge is yet deficient for different reasons. Firstly, despite wide research efforts investigating plastic pollution in oceans, considerable less

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attention has been paid on freshwater systems (Blettler et al., 2018). Nevertheless, this imbalance seems to be reversing in the last years (e.g. Gündoğdu et al., 2018; Battulga et al., 2019; van Wijnen, 2019).

Secondly, research on freshwater plastic pollution have been mainly carried out in industrialized countries (the Global North; Rochman et al., 2015; Blettler et al., 2018). This is not surprising due to the bias in the scientific output between the Global North and the Global South (Guterl, 2012). However, this disparity causes concern, as increasing population levels, rapid urbanization, informal settlements, and the rise in consumption levels have greatly accelerated the solid waste generation rate in the Global South, where waste collection, processing and final disposal is still poor (Minghua et al., 2009; United Nations Human Settlements Programme, 2016).

Thirdly, there is a clear dominance of microplastic over macroplastic studies in freshwater environments worldwide (less than 20% of the total surveys in freshwater systems have been focused on macroplastics; Blettler et al., 2018). Consequently, more macroplastics studies in freshwaters are urgently required since: i) studies estimating the amount of plastic exported from rivers into the ocean are limited due to the scarcity of field-data in rivers (Lebreton et al., 2017; Schmidt et al., 2017); ii) global studies estimating the amount of plastic exported from rivers into the ocean have evidenced a significantly (>100 times) greater input in terms of weight of macroplastics (compared with microplastics, Schmidt et al., 2017); iii) removing macroplastics in rivers (e.g. using artisanal boom barriers) is an effective/low-cost action to avoid plastics reach the ocean but, on the contrary, the same action on microplastic is virtually impossible. Microplastics can be categorized by their source. Primary microplastics are purposefully made to be that size (e.g. microbeads used in cosmetics and personal care products, virgin resin pellets used in plastic manufacturing processes), while secondary microplastics are the result of larger items of plastic breaking down into smaller particles (Weinstein et al. 2016). Studies indicated that wastewater treatment plants (WWTPs) play an important role in releasing primary microplastics to the environment (Ou and Zeng, 2018; Gündoğdu et al., 2018).

Fourthly, the largest rivers in the world (also called mega-rivers) are located in developing countries (see Latrubesse, 2008). The great discharges, basin sizes and poor sanitary conditions of people living in these catchments, potentially increase the amount of plastic debris flowing through mega-rivers to the ocean. However, information about plastic pollution in mega-rivers of developing countries is still very scarce (Pazos et al., 2017; Blettler et al., 2018), even though all the plastic input conveyed by rivers is eventually released into oceans (Morritt et al., 2014) or accumulated in estuaries (Vermeiren et al., 2016).

Fifthly, the ingestion of microplastics by fish, and the associated risks to human health, remain major knowledge gaps (Santos Silva-Cavalcanti et al., 2017), even though the major inland fisheries are located precisely in the most plastic polluted rivers (Lebreton et al., 2017) of the Global South (FAO 2016). The above suggests an urgent need to focus monitoring efforts in the most polluted rivers, specially where inland fisheries are crucial for local consumption and economies, as it is the case with the Paraná River.

Taking into account the rationale outlined above, the objectives of this study were to determine: i) the amount, origin and composition of plastic debris deposited in sediments of a mega-river (Paraná River), ii) the plastic input conveyed by an urban stream joining the Paraná River; iii) quantitative relationship between macro, meso and microplastics in sediments; iv) microplastic ingestion by *Prochilodus lineatus*, an iliophagous fish (that feeds mud containing detritus and associated organisms).

2. Materials and methods

2.1. Study area

La Plata basin is one the ten largest fluvial basins of the world, draining five countries (southern part of Brazil, the northern of Argentina, Bolivia, Uruguay and Paraguay), accounting for 17% of the surface area of the South America and supporting 19 large cities (with a population greater than 100,000 inhabitants). The Paraná River is the largest river of this basin, ranking ninth among the largest rivers of the world, according to its mean annual discharge to the Atlantic Ocean (18,000 $m^3 s^{-1}$; Latrubesse, 2008). However, this river is also one of the world's top-ten rivers at risk due to anthropogenic pressure (Wong et al., 2007).

The study took place near Paraná city (Argentina), located on its eastern shore of the river, with a population of about 300,000 inhabitants. The collection, processing and final disposal of waste of this city is still deficient resulting in strongly polluted urban streams.

We selected three sampling areas in the Paraná River bank sediments: upstream of the city (Escondida beach), in the city (Thompson beach, a municipal public beach), and in an island located in front of the city (Curupí island; Fig. 1). Thompson is a recreational beach influenced by the mouth of a strongly polluted urban river ("Las Viejas" stream) that flows through the Paraná city. Fish were caught in the vicinity of the sampling sites. Due to flow conditions, we expected that the upstream site would be the least polluted, followed by Curupí island, whereas Thompson beach, is influenced by the strongly polluted "Las Viejas" stream crossing the city.

2.2. Sampling

We selected 2 transects of 50 m in length and 3 m wide for the macroplastic survey (Noik and Tuah, 2015) in each sampling area. Transects were selected parallel to the riverbank, randomly chosen, and covering more than a 20% of the shoreline section (Lippiatt et al., 2013). All visible macroplastic items on the surface of each transect were collected by hand.

Plastic debris was sorted according to size and classified as macroplastic (>2.5 cm), mesoplastic (5 mm–2.5 cm), or microplastic (≤5 mm). This classification is currently used by the UNEP (Cheshire et al., 2009), NOAA (Lippiatt et al., 2013) and MSFD (2013).

We collected mesoplastic debris from triplicate samples (1 m^2) randomly located into each macroplastic-transects (after macroplastic being picked up; Lippiatt et al., 2013). Mesoplastics particles were carefully removed from the top 3 cm of sediments of each 1 m^2 quadrat (using stainless steels of 5 mm mesh size to sieved the sediments). In a similar way, we took microplastics samples also per triplicate from the macroplastic-transects but using smaller quadrats (0.25 × 0.25 m x 3 cm depth; Klein et al., 2015). Mesoplastic particles were hand-picked in the field using stainless steels (5 mm mesh size), while microplastic samples were directly transferred to the laboratory for processing.

All sampled (macro and mesoplastics and sediment with microplastics) were transferred to the laboratory for further analyses (see below).

Prochilodus lineatus (locally called "Sábalo") is a dominant detritivorous fish species of great importance for commercial and artisanal fishing (Espínola et al., 2016). For the analysis of fish, we obtained 21 fresh specimens that were caught with gill nets of 14 and 16 cm between opposite knots at the respective sites of the

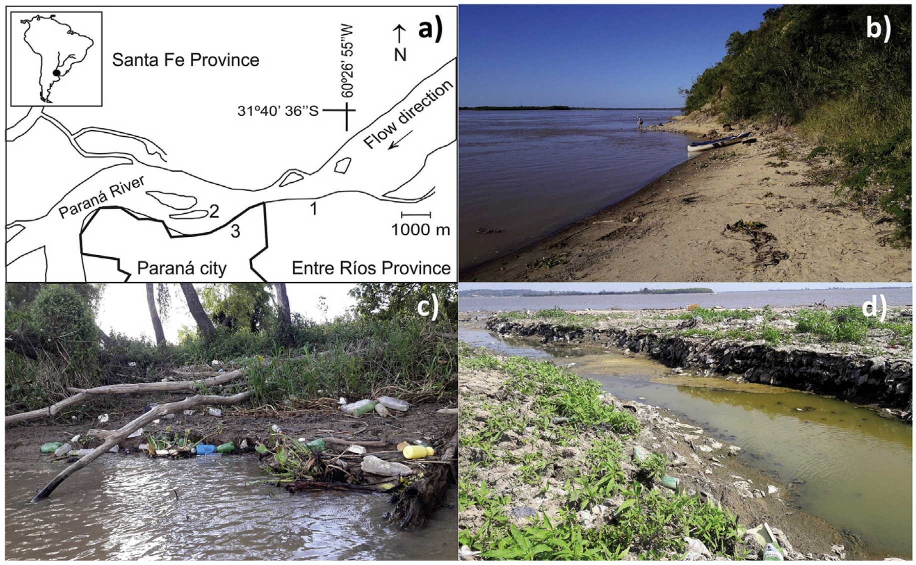

Fig. 1. Location of the Paraná River (study area, Entre Ríos Province, Argentina) in the Global South (a). Escondida beach (b), Curupí island (c), and Thompson beach (at the confluence of Las Viejas urban stream with the Paraná main channel) (d).

study area, respecting local policies. Fish were caught in the early morning hours and transported to the laboratory on ice within 3 h. For each individual, total length (cm) was measured and the body total weight (g) was also determined. Afterward, fish samples were cut open using a scalpel and gastrointestinal tracts were removed and immediately placed in clean glassware in order to minimize the risk of laboratory contamination (Bessa et al., 2018). In addition to the methods described below, we also noted the color of the eaten particles in order to identify potential preferences.

In order to avoid contamination from microplastics, potentially present in the laboratory environment, the use of cotton lab coats, gloves and mask was mandatory. Moreover, glassware and working place were cleaned with solution of ethanol (96%) before starting all experiments in order to conserve a sterile environment. From the beginning of the operations until the observation under the microscope, the samples were covered with aluminium foil.

The organic matter presents in the samples was digested with hydrogen peroxide (H_2O_2) (30%) at 60 °C (Pazos et al., 2017; Jabeen et al., 2017). According to Sujathan et al. (2017), H_2O_2 is an oxidizing agent that no changes or bleach the structure of microplastic particles. According to our environmental principles, all sampling campaigns were performed using kayaks (zero emission and free noise pollution).

2.3. Samples analysis and processing

Macroplastic particles were washed, counted and classified in the laboratory (item by item). The classification accounted for their functional origin (e.g. food wrappers, packaging, beverage bottles, shopping bags, personal care products, etc.) following the NOAA (Lippiatt et al., 2013) and resin composition. The ASTM International Resin Identification Coding System (RIC, 2016) was used to recognize the plastic resin used in manufactured macroplastics (Gasperi et al., 2014). As the later procedure was not always possible to use (sometime this code is lost or not clearly visible), we used a FT–IR Spectrophotometer Shimadzu IR Prestige 21™ to identify the plastic resin (Song et al., 2015).

According to Gündoğdu and Çevik (2017), mesoplastics were counted and classified in: Styrofoam, hard plastic, fishing line, and films.

Microplastic separation was performed following the method proposed by Masura et al. (2015). Thus, full samples were dried at 60 °C per 24hs, weighed and sieved through a stainless steel sieve of 350 μm mesh size using a Retsch™ sieve shaker. The remaining material was transferred to a 1 L beaker for wet 30% peroxide oxidation (H_2O_2), and located on a hot plate set at 60 °C until all organic material digested (Yonkos et al., 2014). After completion, H_2O_2 was washed using distilled water through a 350 μm mesh size. Afterward, a concentrated saline NaCl solution (1.2 g cm^{-3}) was added and strongly stirred for about 1 min (Hidalgo-Ruz et al., 2012). Afterward, the supernatant with floating microplastics was extracted and washed with distilled water for further processing. This last step was repeated as many times as it was needed in order to catch every floating plastic particle.

Microplastics were separated from other materials (present in the supernatant) and classified under a Boeco™ zoom stereo microscope and a Nikon™ binocular microscope (10–40x). We used the criteria suggested by Norén (2007) to identify microplastics. However, items of doubtful origin were analysed with a FT-IR Spectrophotometer in order to confirm (or reject) their plastic composition (Frias et al., 2014; Li et al., 2016). Spectra ranges were set at 4000–400 cm^{-1}, using the IRsolution Agent software. The resulting spectra were directly compared with the reference library databases.

Microplastics were classified in Styrofoam (trademarked brand of closed-cell extruded polystyrene foam), hard plastic, film, fiber and fiber-roll (very large fibers twisted), according to Castañeda et al. (2014) and Gündoğdu and Çevik (2017).

2.4. Data analyses

Tables and figures were created to identify presence, abundance and type of plastic debris in order to compare the sampling sites between each other. Correlations were performed among the different plastic seize ranges. In order to test spatial patterns of similarity in the abundance and type of microplastics, a Canonical Analysis of Principal (CAP) coordinates was performed. The CAP is a constrained ordination analysis that calculates unconstrained principal coordinate axes followed by canonical discriminant analysis on the principal coordinates to maximize the separation between predefined groups (Anderson, 2004). The Bray-Curtis dissimilarity index and 999 permutations were the parameters selected in this procedure. Subsequent one-way Permutational Multivariate Analyses of Variance (PERMANOVA) (Anderson, 2001) was conducted to determine differences between scores of the CAP Axis 1.

Statistical analyses were carried out using the CAP software Version 1.0 (Anderson, 2004) and the MULTIV software, version 2.4.2 (Pillar, 2004), with a statistical significance level was $p < 0.05$.

Table 1
Type (origin/use), density per transect (150 m^2), standard deviation, abundance (%) and resin composition of macroplastic debris (total and per sampling site). Where, HDPE: high-density polyethylene; LDPE: low-density polyethylene; PP: Polypropylene; PS: Polystyrene; EPS: Expanded polystyrene; PET: Polyethylene terephthalate; Nylon: dry polyamide; PE: Polyethylene; PVC: Polyvinyl chloride.

Type of debris	N° of items per transect (150 m^2) and Standard Deviation	%	Resin
Bag	166.2 ± 252.1	48.75	HDPE, LDPE
Foodwrapper	68.3 ± 110.1	20.05	PP, PS
Styrofoam	35.5 ± 61.5	10.42	EPS
Beverage bottle	30.7 ± 31.2	9.00	PET
Fishing line	8.5 ± 15.7	2.49	Nylon
Bottle cap	4.7 ± 6.3	1.37	PP
Food containers (hard)	3.3 ± 8.2	0.98	PS, PET
Cleaning bottle	3.2 ± 4	0.93	HDPE, PET
Sanitary napkin	1.7 ± 4.1	0.49	PP, PE
Household appliances	1 ± 0.4	0.29	Undetermined
Personal care container	0.8 ± 2	0.24	PP, HDPE, PET, PDPE, Varies
Strapping band	0.8 ± 2	0.24	Polyester, PP
Cloth	0.3 ± 0.5	0.10	Polyester
Bottle label	0.2 ± 0.4	0.05	PET, PP, PVC
Straw	0.2 ± 0.4	0.05	PP
Diaper	0.2 ± 0.4	0.05	PP, PET
Cigarette butt	0.2 ± 0.4	0.05	Cellulose acetate
Others	15.2 ± 19.2	4.45	Undetermined
Total	340.8	100	
Site			
Escondida	52 ± 42.4	5.1	
Curupí	190 ± 77.1	18.6	
Thompson	780 ± 14.1	76.3	

3. Results

3.1. Macroplastics

We recorded a total of 18 categories of macroplastic debris (based on the NOAA's classification; Lippiatt et al., 2013); being bag, foodwrapper, Styrofoam and beverage bottle the most abundant particles, representing almost the 80% of the total (Table 1).

The three sampling sites have strong differences in amount (number of items) and type of macroplastic debris (Fig. 2a). Thus, Escondida beach (4 km upstream Paraná city) showed the lower values (52 macro-items per transect; 150 m^2), with a heterogeneous composition of plastic types (13 different categories) but dominated by fishing lines (23 items). The Curupí island (in front of the Paraná city), was dominated by only 2 types of macroplastics: beverage bottles (81) and Styrofoam fragments (99). Finally, the Thompson beach (slightly downstream to the Las Viejas outlet) showed a clear dominance of shopping bags (490; many different colours and textures) and food wrappers (202.5), having the highest amount of plastics: 757.5 items per transect (i.e. 5.05 macroplastic particles per m^2), 14 times more than the Escondida beach. By far, the most abundant plastic resins were HDPE, LDPE, PP and PS in the Thompson beach, EPS and PET in the Curupí island and Nylon in the Escondida beach. Cellulose acetate, Polyester and PVC resins were found at low densities.

3.2. Mesoplastics

In contrast to macroplastics, mesoplastics had the highest abundance in the Escondida beach (55.6 items m^{-2}), followed by Curupí island (35.5 items m^{-2}) and Thompson beach (only 18.5 particles per m^2; Fig. 2b). The average abundance of mesoplastic was close to 46 items m^{-2}, being foam plastic (Styrofoam) the dominant category (41.1 items m^{-2}) (Table 2).

3.3. Microplastics

Films and fibers were the dominant items in the microplastic samples (Table 3). An average of 4654 microplastic fragments (per m^2) was found in shoreline sediments of the three sampling (beaches and island). An average of 12687 micro-particles m^{-2} (81% of the total) were recorded in the Thompson beach, but only 131 in the Curupí island (Fig. 2c). Microplastic film and fibber were extremely abundant in the Thompson beach.

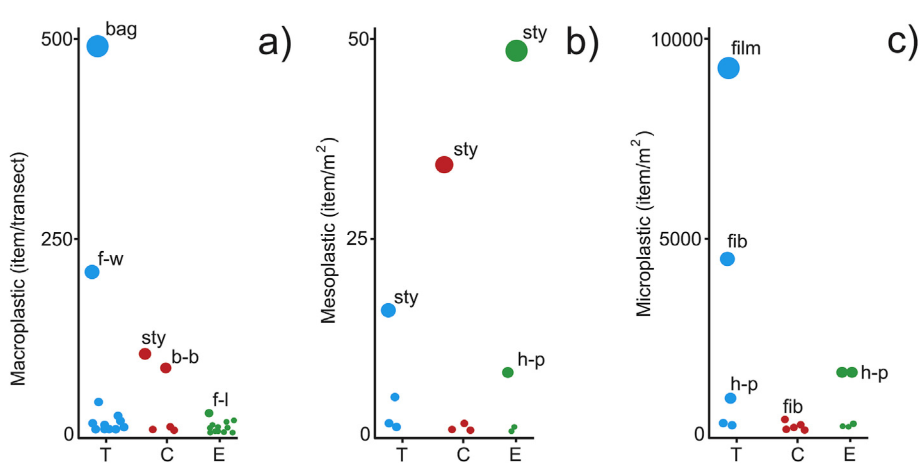

Fig. 2. Bubble chart showing macro- (a), meso- (b) and microplastic (c) densities at each sampling area. Where: f-w: foodwrapper, sty: Styrofoam, b-b: beverage bottle, fishing-line, h-p: hard-plastic piece, fib: fibber.

Table 2
Type, density (m^2), standard deviation, and abundance (%) of mesoplastic debris per sampling site.

Mesoplastic Type	Escondida	Curupí	Thompson	Standard deviation	%
Styrofoam	47.8	35.5	16	48.3	89.3
Hard plastics	7.5	0	2.5	7.6	10
Fishing line	0.2	0	0	0.2	0.2
Cassette tape	0.2	0	0	0.2	0.2
Total (mean)	55.6	35.5	18.5	18.6	100

The CAP (and subsequent PERMANOVA) showed significant differences in abundance and type of microplastics between the three beaches (sampling sites) (p-values= <0.003; Sum of squares (Q) within groups = 2.829) (Fig. 3).

Table 4 shows that the density values of the size classes (macro, meso and microplastic) were not surrogate of each other (no correlations were detected). While some weak tendencies could be detected (ex.: high concentration values of macro and microplastics in the Thompson beach), they were not statistically significant. Particularly, the mesoplastic abundance showed a completely independent tendency. For ex.: lowest values of macroplastic were found in the Escondida beach, but mesoplastic showed the highest concentration in the same beach. While the highest concentrations of macro- and microplastics were found in the Thompson beach, the mesoplastic concentration there was the lowest one.

Table 4
Correlations among the different plastic seize ranges.

	r^2	*p* value
Macro- *vs.* meso-p	0.006	0.85
Meso- *vs.* micro-p	0.022	0.72
Micro- *vs.* macro-p	0.199	0.27

3.4. Fish ingestion

All fish were contaminated with at least one microplastic. The number of items recorded in the digestive tracts of adult *P. lineatus* averaged 9.9 microplastic particles, The maximum value of microplastic particles recorded in an individual was 27 (Fig. 4). Particle sizes ranged between 0.5 and 3 mm and recorded colours were blue (most of them), black, yellow, red and transparent.

4. Discussion

4.1. Massive plastic concentration: geo-political issues and societies

Macroplastic materials are the most visible form of plastic pollution. Blettler et al. (2017) reported an average of 172.5 macroplastic items per transect of 150 m^2 (~1.15 items m^2) in a floodplain lake of the Paraná River, located only 18 km from our

Table 3
Type, density (m^2), standard deviation, and abundance (%) of microplastic debris per sampling site.

	Escondida	Curupí	Thompson	Standard deviation	%	Category
Fiber	1431.4	90	4466.9	1899.6	33.1	Primary
Hard plastics	1424.2	18.8	421.7	51.8	0.9	Secondary
Styrofoam	33.2	11.3	36.2	2645.4	17.5	Secondary
Film	0	0.8	8953.5	6772.3	48.2	Secondary
Fiber-roll	0	0	72.9	54.5	0.4	Primary
Total (mean)	2899	131	12687	8548.1	100	

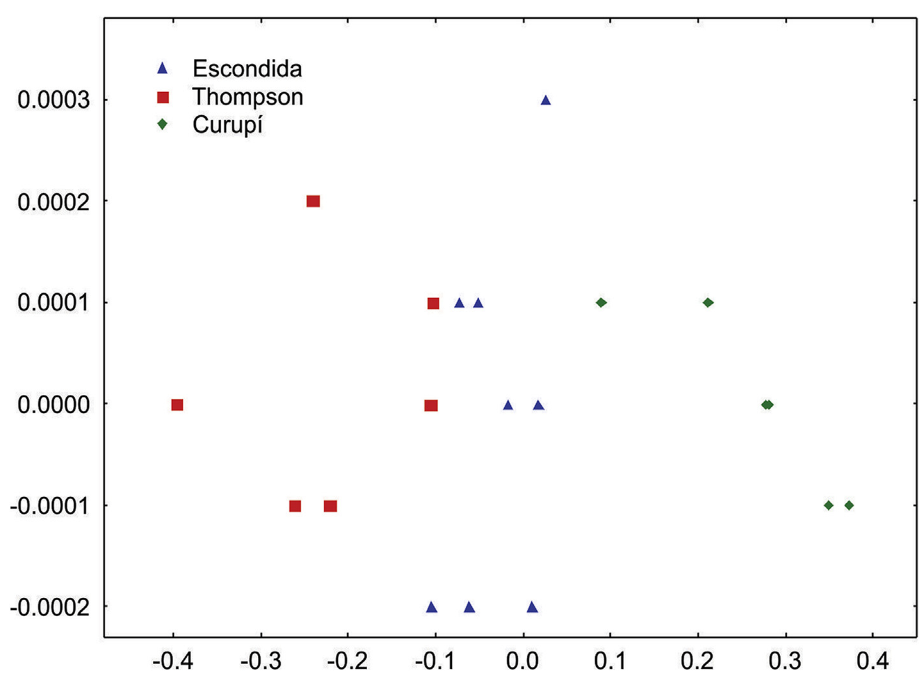

Fig. 3. Ordination plot of the Canonical Analysis of Principal coordinates (CAP) showing significant differences in abundance and type of microplastics between the three sampling sites (Escondida beach, Thompson beach, Curupí island).

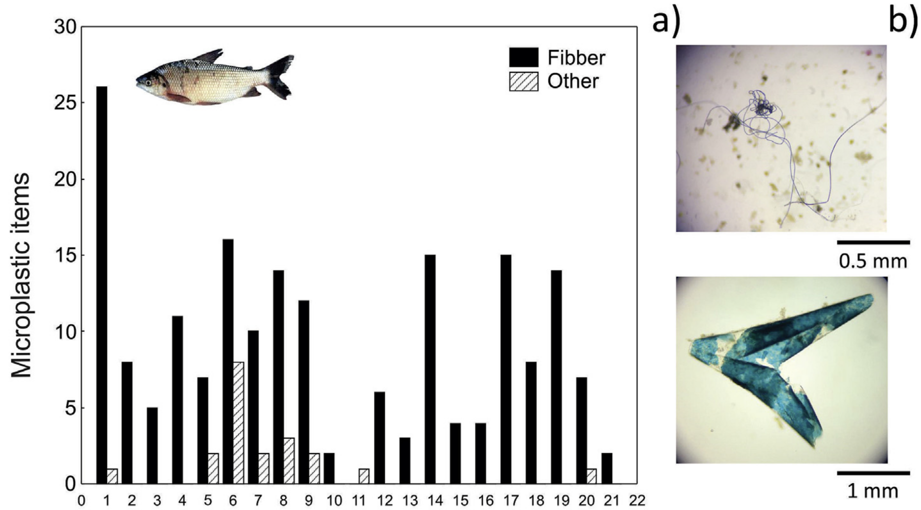

Fig. 4. Microplastic particles (fibers and others) found in the digestive tracts of *P. lineatus*. Number of items (a), fibers and a piece of plastic film (b).

sampling area. In the present study, we found almost twice that amount: 340.8 macroplastics per 150 m^2 (~2.27 m^2).

While several studies on macroplastics have been performed in water surface of rivers (Gasperi et al., 2014; Faure et al., 2015; Baldwin et al., 2016; Lahens et al., 2018) and lakes (Faure et al., 2015), macroplastic studies in riverine sediments are still scare, especially for beaches. Some examples include Imhof et al. (2013) in the Garda lake (Italy) and Faure et al. (2015) in 6 lakes of Switzerland. However, direct comparison with the present study are unfeasible since these authors considered macroplastics as the particles higher than 5 mm (including mesoplastic size).

The great amount of macroplastic debris recorded in the Thompson beach and Curupí island, as well as the origin of them (household waste, Table 1), suggest a deficient waste collection, processing and final disposal in the Paraná city. Waste management is one of the key environmental issues concerning urban hydrosystems on a global scale, however, in the Global South it still remains strongly based on uncontrolled dumping and/or littering (Guerrero et al., 2013). As a result, serious environmental problems (Al-Khatib et al., 2010) and increasing plastic pollution (Battulga et al., 2019) occur, particularly in freshwater systems. Municipalities in low-income countries are spending lower proportion of their budgets on waste management, and yet over 90% of waste in low-income countries is still openly dumped (Kaza et al., 2018). In addition, increasing population levels and the rise in consumption levels have greatly accelerated the solid waste generation rate in Argentina (waste generation rates: 1.14 kg/capita/day; Kaza et al., 2018). The present study shows, in part, this global trend.

Most of the macroplastics recorded in the present research were shopping bags, followed by food wrappers and foam packaging (almost 80%; Table 1). The first communities to embrace the anti-plastic bag norm were in the Global South, with those in the Global North only doing so much more recently (Clapp and Swanston, 2009). However, an anti-plastic-bag municipal ordinance was not adopted in the Paraná city before 2017.

Results from available microplastics studies in freshwater systems are extremely variable according to the used methodology used (e.g. grab sampler, sediment core, manta net, pump, etc), size range reported (including nanoplastic), reporting unit (e.g. m^2, m^3, l, kg), environment (river, lake, reservoir, estuary, sewage, etc), and sampling compartment (water surface or column, bottom or beach sediment, etc). As a result, comparisons between worldwide studies are very difficult. We found an average of 5239 microplastics m^{-2} (size range: 0.35–5 mm) in bank sediments of the Paraná River, ranging from only 75 to a maximum of 34443 microplastics m^{-2} (Table 3). Castañeda et al. (2014) found about 13832 m^{-2} polyethylene microbeads, retained by a 0.5 mm sieve, from industrial effluents in the St. Lawrence River sediments (Canada). Klein et al. (2015) have record about 228–3763 microparticles kg^{-1} in shore sediments of the Rhine and Main rivers in Germany (microplastic size: 0.2–5 mm). Moreover, Su et al. (2016) have reported a range of 15–1600 microplastics l^{-1} (>0.3 mm) in the Middle-Lower Yangtze River (China), Wang et al. (2016) recorded 178-544 microplastics l^{-1} (<5 mm) in the Beijiang River sediments, and Peng et al. (2017) found 410-1600 microplastics kg^{-1} (0.05–5 mm) in some rivers of Shanghai, most of them fragments, spheres and fibers.

Blettler et al. (2017), using the same methodology as the present study, have recorded a much lower average of 704 microplastics m^{-2} (size range: 0.35–5 mm) in beach sediments of lentic environments of the Paraná River (a floodplain lake located 18 km from the sampling area of the present study). Xiong et al. (2018) reported 50-1292 microplastics m^{-2} (>0.1 mm) in the Qinghai Lake (China); most of them were films, fibers and foams.

In spite of the limitations and weaknesses of the above comparisons (i.e. different size ranges, units, environments), available information suggest a significant microplastic pollution present in sediments of the Paraná River.

The variation of microplastics abundance and type between sampling sites was statistically significant (Fig. 4), showing a clear differentiation per sampling beach. Thompsons beach showed the highest concentration of microplastics, while Escondida revealed the most heterogeneous distribution (sampling stations ranged from low to high microplastic concentration).

Microplastic can occur either in a primary (beads) or secondary form (originating from the breakdown of larger plastic items; Cole et al., 2011). The relative importance of primary versus secondary sources of microplastics is still unknown. We found both of them, but the secondary ones were considerably more abundant (Table 3).

Particular attention should be paid to synthetic clothes, which are an important source of fibers via washing (Conkle et al., 2018). In our study, fiber was the only primary microplastic (Cole et al., 2013) recorded. However, it should be noted that some authors consider fiber as secondary (e.g.: Dris et al., 2015). Other primary microplastics such as microbeads, capsules or pellets (used in cosmetics and personal care products, industrial scrubbers used for abrasive blast cleaning and virgin pellets used in plastic manufacturing processes, respectively) were absent. Similar lack of microbeads was observed in the Yangtze River (Zhang et al., 2015) and the Three Gorges Reservoir (Zhang et al., 2017) in China, the Saigon River in Vietnam (Lahens et al., 2018), and the Paraná River estuary in Argentina (Pazos et al., 2018). Nevertheless, a great presence of microbeads was observed in the Rhine and St. Lawrence Rivers (Mani et al., 2015 and Castañeda et al., 2014, respectively) and in Laurentian Great Lakes (Eriksen et al., 2013). In some countries benefiting from advanced waste treatment facilities (mainly in Europe and North of America), secondary microplastics releases are even lower than primary microplastics (Gouin et al., 2015). Losses of primary microplastics can occur during the production, transport or recycling stages of plastics, or during the use phase of products containing microplastic (e.g. microbeads originated from facial cleansers widely used in developed nations; Napper et al., 2015; Gouin et al., 2015). This contrasts with secondary microplastics that mostly originate from mismanaged waste during the disposal of products containing plastics (Boucher and Friot, 2017). The absence of microbeads in the Paraná River system could be explained by these differences in consumer habits and waste management between societies and countries. Herein, almost 50% of the recorded microplastics were film particles (as a secondary product of advanced bag breakdown process), 33.1% fibers (used in textiles) and 18.7% resulting from larger particles of plastic of uncertain origin breaking down into smaller items (probably beverage bottle, foodwrapper and foams) (Table 3). In contrast, other studies in rivers from developing countries have reported a dominance of microplastic fibers (Zhang et al., 2015; Lahens et al., 2018), even in the Paraná River estuary (Pazos et al., 2018).

The variable ratios between macro- or mesoplastics in our study have shown that these data cannot serve as surrogates for microplastics monitoring (Table 4). This is important since surveys of macroplastics debris can be easily conducted by volunteers, who have played important roles in many debris monitoring programs (Ribic et al., 2012).

4.2. Role of urban streams in plastic dissemination

Urban rivers and streams suffer from multiple interactive stressors, especially in the Global South (Wang et al., 2012; Wantzen et al., 2019). In this study, Las Viejas urban stream seems to play a crucial role transporting huge amounts of waste plastics and depositing them into the Thompson beach, immediately downstream to the confluence with the Paraná River (Fig. 1d). This sampling area showed the highest concentration of macro and microplastic debris (Figs. 2 and 4). Las Viejas stream flows all through the Paraná city, concentrating and transporting the municipal solid waste improperly managed. According to Xu et al. (2019) the development of sewer systems has not caught up with the urbanization speed in developing countries, with serious consequences for urban river water quality. Thus, many urban rivers become the end points of plastic pollution (McCormick et al., 2014, 2016). In the same way as rains and severe floods can dramatically increase the plastic levels in the sea (Gündoğdu et al., 2018), it is highly probable that the same phenomenon operates in urban streams discharging to large river systems.

On the other side, the Curupí island showed an average of 190 macroplastics per transect (against 780 in the Thompson and only 52 in the Escondida beach; Table 1). This sampling site was dominated by two domestic items: beverage bottles and foam packaging fragments (Styrofoam; Fig. 2). We hypothesize that these plastics arrived from Las Viejas stream. Floating waste is transported by the Paraná River current and dominant southern winds unto the Curupí island shores. This process could be facilitated by the high buoyancy of these items (EPS density: 11-32 kg m^{-3}; while density of PET is 950 kg m^{-3} bottles initially float due to the air trapped inside). Otherwise, shopping bags and food wrappers (most abundant items in the Thompson beach) were not recorded in the island which is, probably, related to their low buoyance (density of HDPE: 950 kg m^{-3}; LDPE: 917-930 kg m^{-3}, PP: 946 kg m^{-3}; PS: 1066 kg m^{-3}).

Finally, there are no urban river confluences in the Escondida beach, which was the least polluted sampling area. This beach showed a completely different plastic debris composition. While shopping bags, Styrofoam and beverage bottles were present, the dominant item was fishing line. It suggests that the main impact is given by the beach users, most of them artisanal and sports fishermen, and not by municipal waste poorly treated coming from large cities upstream.

The most common plastic polymers recorded in this study were HDPE, LPDE, PP, PS and EPS, which can be very harmful to wild fauna (Kyaw et al., 2012). Moreover, PP and PS have been extensively recorded in food wrappers particles (Table 1). Finally, EPS (often referred as Styrofoam™) products (takeout containers, dispensable cups, foam trays, etc) were widespread found in our study (Table 1). EPS is commonly reported as one of the top items of debris recovered from shorelines and beaches worldwide (Lee et al., 2013; Ocean Conservancy, 2017). As a result, EPS products are now discussed for a ban in several countries (UNEP 2018). In the present study, EPS was the most abundant mesoplastic debris (almost 90%; Table 2). Zbyszewski et al. (2014) reported a similar proportion in mesoplastics from the Great Lakes.

4.3. Ingestion of plastic by fish and potential impacts

A recent study revealed that plastic ingestion has been reported in 427 fish species, from more than 20 countries around the globe (Azevedo-Santos, 2019), causing internal blockages and injury to the digestive tract of fish (Cannon et al., 2016; Nadal et al., 2016). We recorded microplastics in the digestive tract of 100% of the sampled *P. lineatus* specimens, corroborating a similar study in the Paraná River estuary (Pazos et al., 2017). The latter could be explained from the detritivorous feeding strategy of this species and the high amount of microplastics recorded in the study area. Thus, the occurrence frequency of microplastics in fish from Paraná River seems to be higher than in other South American rivers. For example, in the Amazon estuary and northern coast of Brazil microplastics were found in 13.8% of digestive tracts examined (Pegado et al., 2018), 23% and 13.4% in the Goiana estuary (Possatto et al., 2011 and Ramos et al., 2012, respectively). However, we recognize that the low number of specimens studied here does not allow generalizations.

In our study, most of the recorded microplastics in fish were fibers (90%). In agreement, several studies worldwide have also reported greater number of ingested fibers compared to other microplastic types (Neves et al., 2015; Bellas et al., 2016; Nadal et al., 2016; Pazos et al., 2017). The reasoning behind the dominance of fibers is the diverse nature of this microplastic type, which may originate from the degradation of clothing items, furniture and fishing gear. Indeed, washing (through a washing machine) a single item of synthetic clothing resulted in the release of about 2000 microfibers (Browne et al., 2011; Carney Almroth et al. 2018). Mesoplastics ingested by fish were not recorded in this study. In fact, this range size has been scarcely recorded in fish digestive tracts (Jabeen et al., 2017).

5. Conclusions

1. The recorded plastic debris concentration (macro, meso and microplastics) was several times higher than the values previously reported in the Paraná River floodplain. Comparisons with other studies worldwide are still difficult, since methodological protocols are not yet standardized; however, they suggest massive pollution levels in this mega-river of South America.
2. Macroplastics recorded herein have a domestic origin (shopping bags, food wrappers, beverage bottles and packaging foam fragments), suggesting an inadequate waste collection, processing and final disposal in the region, which is regrettably recurrent in the Global South. The further research must not overlook macroplastics in this geopolitical region, particularly if reliable estimates of global plastic waste entering to the ocean from rivers are intended.
3. Secondary microplastics (originated from the breakdown of larger plastic items) were more abundant than primary ones (manufactured as microbeads, capsules, pellets used in industry). Microbeads (commonly found in industrialized regions) were absent in the Paraná River. This finding contrasts with studies performed in freshwater environments of developed countries, suggesting a difference in consumer habits and levels of industrialization between societies and economies from the developed and developing world.
4. Most of the recorded plastic debris proceed from a highly polluted urban stream, which runs through the Paraná city. Urban rivers, particularly in the Global South, are vulnerable to different urban processes and activities that cause pollution and degradation of the water ecosystem.
5. We recorded microplastic particles in the digestive tract of 100% of *P. lineatus* specimens, most of them were fibers. While we recognize the low number of collected fish, this finding evidenced that microplastics have penetrated in the aquatic food webs and ecological niches in the Paraná River, reinforcing the necessity of more studies.
6. Contrary to our expectations, the macroplastic or mesoplastic items would not serve as surrogates for microplastic surveys (and vice versa), suggesting that all plastic debris sizes should be considered in further studies.

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Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.envpol.2019.113348.

References

Al-Khatib, I.A., Monou, M., Zahra, A.S.F.A., Shaseen, H.Q., Kassinos, D., 2010. Solid Waste characterization, quantification and management practices in developing countries. A case study: Nablus district-Palestine. J. Environ. Manag. 91, 1131–1138.
Anderson, M., 2001. Permutation tests for univariate or multivariate analysis of variance and regression, Can. J. Fish. Aquat. Sci. 58, 626–639.
Anderson, M., 2004. CAP: A FORTRAN Computer Program for Canonical Analysis of Principal Coordinates. New Zealand: Department of Statistics. University of Auckland.
Azevedo-Santos M., V., Gonçalves, G. R. L., Manoel, P. S., Andrade, M. C., Lima, F. P., Pelicice, F. M., 2019. Plastic ingestion by fish: A global assessment. Environmental Pollution 255 (112994).
Baldwin, A., Corsi, S., Mason, S., 2016. Plastic debris in 29 great lakes tributaries: relations to watershed attributes and hydrology. Environ. Sci. Technol. 50, 10377–10385.
Battulga, B., Kawahigashi, M., Oyuntsetseg, B., 2019. Distribution and composition of plastic debris along the river shore in the Selenga River basin in Mongolia. Environ. Sci. Pollut. Res. 26, 14059–14072.
Beaumont, N.J., Aanesen, M., Austen, M.C., Börger, T., Clark, J.R., Cole, M., Hooper, T., Lindeque, P.K., Pascoe, C., Wyles, K.J., 2019. Global ecological, social and economic impacts of marine plastic. Mar. Pollut. Bull. 142, 189–195.
Bellas, J., Martínez-Armental, J., Martínez-Cámara, A., Besada, V., Martínez-Gomes, C., 2016. Ingestion of microplastics by demersal fish from the Spanish Atlantic and Mediterranean coasts. Mar. Pollut. Bull. 109, 55–60.
Bessa, F., Barría, P., Neto, J.M., Frias, J.P., Otero, V., Sobral, P., Marques, J.C., 2018. Occurrence of microplastics in commercial fish from a natural estuarine environment. Mar. Pollut. Bull. 128, 575–584.
Blettler, M., Ulla, M.A., Rabuffetti, A.P., Garello, N., 2017. Plastic pollution in freshwater ecosystems: macro-, meso-, and microplastic debris in a floodplain lake. Environ. Monit. Assess. https://doi.org/10.1007/s10661-017-6305-8, 189,581.
Blettler, M., Abrial, E., Khan, F., Sivri, N., Espinola, L., 2018. Freshwater plastic pollution: recognizing research biases and identifying knowledge gaps. Water Res. 143, 416–424.
Boucher, J., Friot, D., 2017. Primary Microplastics in the Oceans: A Global Evaluation of Sources. Gland. IUCN, Switzerland, p. 43. DOI: dx.doi.org/10.2305/IUCN.CH.2017.01.en.
Browne, M.A., Crump, P., Niven, S.J., Teuten, E., Tonkin, A., Galloway, T.-, Thompson, R., 2011. Accumulation of microplastic on shorelines worldwide: sources and sinks. Environ. Sci. Technol. 45, 9175–9179.
Cannon, S.M.E., Lavers, J.L., Figueireda, B., 2016. Plastic ingestion by fish in the Southern Hemisphere: a baseline study and review of methods. Mar. Pollut. Bull. 107, 286–291.
Carney Almroth, B., Åström, L., Roslund, S., Petersson, H., Johansson, M., Persson, N.-K., 2018. Quantifying shedding of synthetic fibers from textiles; a source of microplastics released into the environment. Environ. Sci. Pollut. Res. 25 (1191) https://doi.org/10.1007/s11356-017-0528-7.
Castañeda, R., Avlijas, S., Ricciardi, A., 2014. Microplastic pollution in St. Lawrence River sediments. Can. J. Fish. Aquat. Sci. 71, 1767–1771.
Cheshire, A.C., Adler, E., Barbière, J., Cohen, Y., Evans, S., Jarayabhand, S., Jeftic, L., Jung, R.T., Kinsey, S., Kusui, E.T., Lavine, I., Manyara, P., Oosterbaan, L., Pereira, M.A., Sheavly, S., Tkalin, A., Varadarajan, S., Wenneker, B., Westphalen, G., 2009. UNEP/IOC Guidelines on Survey and Monitoring of Marine Litter, p. 120. UNEP Regional Seas Reports and Studies No. 186, IOC Technical Series No. 83.
Clapp, J., Swanston, L., 2009. Doing away with plastic shopping bags: international patterns of norm emergence and policy implementation. Environ. Pol. 18, 315–332.
Cole, M., Lindeque, P., Halsband, C., Galloway, T.S., 2011. Microplastics as contaminants in the marine environment: a review. Mar. Pollut. Bull. 62, 2588–2597.
Cole, M., Lindeque, P., Fileman, E., Halsband, C., Goodhead, R., Moger, J., Galloway, T.S., 2013. Microplastic ingestion by zooplankton. Environ. Sci. Technol. 47, 6646–6655.
Conkle, J.L., Báez Del Valle, C.D., Turner, J.W., 2018. Are we underestimating microplastic contamination in aquatic environments? J. Environ. Manag. 61, 1–8.
Ocean Conservancy, 2017. International coastal cleanup report. Available online at. https://oceanconservancy.org/blog/2017/06/05/results-internationalcoastal-cleanup/.
Dris, R., Gasperi, J., Rocher, V., Saad, M., Renault, N., Tassin, B., 2015. Microplastic contamination in an urban area: a case study in Greater Paris. CSIRO Publ. Environ. Chem. 2015. ff10.1071/EN14167f.
Eriksen, M., Mason, S., Wilson, S., Box, C., Zellers, A., Edwards, W., Farley, H., Amato, S., 2013. Microplastic pollution in the surface waters of the Laurentian great lakes. Mar. Pollut. Bull. 77, 177–182.
Espínola, L.A., Rabuffetti, A.P., Abril, E., Amsler, M.L., Blettler, M.C., Paira, A.R., Simoes, N.R., Santos, L.N., 2016. Response of fish assemblage structure to changing flood and flow pulses in a large subtropical river. Mar. Freshw. Res. 68, 319–330.
FAO (Food and Agriculture Organization), 2016. The State of World Fisheries and Aquaculture 2016. in: Contributing to Food Security and Nutrition for All, p. 200. Rome, ISBN 978-92-5-109185-2.
Faure, F., Demars, C., Wieser, O., Kunz, M., de Alencastro, L., 2015. Plastic pollution in Swiss surface waters: nature and concentrations, interaction with pollutants. Environ. Chem. 12, 582–591.
Frias, J.P.G.L., Otero, V., Sobral, P., 2014. Evidence of microplastics in samples of zooplankton from Portuguese coastal waters. Mar. Environ. Res. 95, 89–95.
Galloway, T.S., Cole, M., Lewis, C., 2017. Interactions of microplastic debris throughout the marine ecosystem. Nat. Ecol. Evol. 1, 116. https://doi.org/10.1038/s41559-017-0116.
Gasperi, J., Dris, R., Bonin, T., Rocher, V., Tassin, B., 2014. Assessment of floating plastic debris in surface water along the Seine River. Environ. Pollut. 195, 163–166.
Gouin, T., Avalos, J., Brunning, I., Brzuska, K., de Graaf, J., Kaumanns, J., Konong, T., Meyberg, M., Rettinger, K., Schlatter, H., 2015. Use of micro-plastic beads in cosmetic products in Europe and their estimated emissions to the North Sea

environment. SOFW J. (Seifen Ole Fette Wachse) 141, 40–46.
Guerrero, L.A., Maas, G., Hogland, W., 2013. Solid waste management challenges for cities in developing countries. Waste Manag. 33, 220–232.
Gündoğdu, S., Çevik, C., 2017. Micro- and mesoplastics in Northeast Levantine coast of Turkey: the preliminary results from surface samples. Mar. Environ. Res. 118, 341–347.
Gündoğdu, S., Çevik, C., Ayat, B., Aydoğan, B., Karaca, S., 2018. How microplastics quantities increase with flood events? An example from Mersin Bay NE Levantine coast of Turkey. Environ. Pollut. 239, 342–350.
Guterl, F., 2012. Scorecard: the world's best countries in science. Sci. Am. 307, 44–45.
Hidalgo-Ruz, V., Gutow, L., Thompson, R.C., Thiel, M., 2012. Microplastics in the marine environment: a review of the methods used for identification and quantification. Environ. Sci. Technol. 46, 3060–3075.
Imhof, H., Ivleva, N., Schmid, J., Niessner, R., Laforsch, C., 2013. Contamination of beach sediments of a subalpine lake with microplastic particles. Curr. Biol. 23, 867–868.
Jabeen, K., Su, L., Li, J., Yang, D., Tong, C., Mu, J., Shi, H., 2017. Microplastics and mesoplastics in fish from coastal and fresh waters of China. Environ. Pollut. 221, 141–149.
Kaza, S., Yao, L., Bhada-Tata, P., Van Woerden, F., 2018. What a waste 2.0: a global snapshot of solid waste management to 2050. World Bank Group. https://doi.org/10.1596/978-1-4648-1329-0.
Klein, S., Worch, E., Knepper, T.P., 2015. Occurrence and spatial distribution of microplastics in river shore sediments of the rhine-main area in Germany. Environ. Sci. Technol. 49, 6070–6076.
Kooi, M., Besseling, E., Kroeze, C., van Wezel, A.P., Koelmans, A.A., 2018. Modeling the fate and transport of plastic debris in freshwaters: review and guidance. In: Wagner, M., Lambert, S. (Eds.), Freshwater Microplastics. The Handbook of Environmental Chemistry, vol. 58. Springer, Cham.
Kyaw, B.M., Champakalakshmi, R., Sakharkar, M.K., Lim, C.S., Sakharkar, K.R., 2012. Biodegradation of low density polythene (LDPE) by Pseudomonas Species. Indian J. Microbiol. 52, 411–419.
Lahens, L., Strady, E., Kieu-Le, T.C., Dris, R., Boukerma, K., Rinnert, E., Gasperi, J., Tassin, B., 2018. Macroplastic and microplastic contamination assessment of a tropical river (Saigon River, Vietnam) transversed by a developing megacity. Environ. Pollut. 236, 661–671.
Latrubesse, E., 2008. Patterns of anabranching channels: the ultimate end-member adjustment of mega rivers. Geomorphology 101, 130–145.
Lebreton, L., van der Zwet, J., Damsteeg, J.-W., Slat, B., Andrady, A., Reisser, J., 2017. River plastic emissions to the world's oceans. Nat. Commun. 7 (8), 15611. https://doi.org/10.1038/ncomms15611.
Lee, J., Hong, S., Song, Y.K., Hong, S.H., Jang, Y.C., Jang, M., Heo, N.W., Han, G.M., Lee, M.J., Kang, D., Shim, W.J., 2013. Relationships among the abundances of plastic debris in different size classes on beaches in South Korea. Mar. Pollut. Bull. 77, 349–354.
Li, J., Qu, X., Su, L., Zhang, W., Yang, D., Kolandhasamy, P., Li, D., Shi, H., 2016. Microplastics in mussels along the coastal waters of China. Environ. Pollut. 214, 177–184.
Lippiatt, S., Opfer, S., Arthur, C., 2013. Marine Debris Monitoring and Assessment. NOAA technical memorandum NOS-OR&R-46.
Mani, T., Hauk, A., Walter, U., Burkhardt-Holm, P., 2015. Microplastics profile along the Rhine River. Sci. Rep. 5, 17988.
Masura, J., Baker, J., Foster, G., Arthur, C., 2015. Laboratory Methods for the Analysis of Microplastics in the Marine Environment: Recommendations for Quantifying Synthetic Particles in Waters and Sediments. NOAA Technical Memorandum NOS-OR&R-R-48.
McCormick, A., Hoellein, T.J., Mason, S.A., Schluep, J., Kelly, J.J., 2014. Microplastic is an abundant and distinct microbial habitat in an urban river. Environ. Sci. Technol. 48, 11863–11871.
McCormick, A.R., Hoellein, T.J., London, M.G., Hittie, J., Scott, J.W., Kelly, J.J., 2016. Microplastic in surface waters of urban rivers: concentration, sources, and associated bacterial assemblages. Ecosphere 7 (11), e01556. https://doi.org/10.1002/ecs2.1556.
Minghua, Z., Xiumin, F., Rovetta, A., Qichang, H., Vicentini, F., Bingkai, L., Giusti, A., Yi, L., 2009. Municipal solid waste management in Pudong new area, China. J. Waste Manag. 29, 1227–1233.
Morritt, D., Stefanoudis, P.V., Pearce, D., Crimmen, O.A., Clark, P.F., 2014. Plastic in the Thames: a river runs through it. Mar. Pollut. Bull. 78, 196–200.
MSFD, Technical Subgroup on Marine Litter, 2013. Guidance on monitoring of marine litter in European seas. Joint Res. Centre Sci. Policy Rep. 128. European Commission.
Nadal, M.A., Alomar, C., Deudero, S., 2016. High levels of microplastic ingestion by the semipelagic fish bogue *Boops boops* (L.) around the Balearic Islands. Environ. Pollut. 214, 517–523.
Napper, I.E., Bakir, A., Rowland, S.J., Thompson, R.C., 2015. Characterisation, quantity and sorptive properties of microplastics extracted from cosmetics. Mar. Pollut. Bull. 99, 178–185.
Neves, D., Sobral, P., Ferreira, J.L., Pereira, T., 2015. Ingestion of microplastics by commercial fish off the Portuguese coast. Mar. Pollut. Bull. 101, 119–126.
Noik, V.J., Tuah, P.M., 2015. A first survey on the abundance of plastics fragments and particles on two sandy beaches in Kuching, sarawak, Malaysia. IOP Conf. Series: Mater. Sci. Eng. 78 012035. https://doi.org/10.1088/1757–1899X/78/1/012035.
Norén, F., 2007. Small Plastic Particles in Coastal Swedish waters. KIMO report.
Ou, H., Zeng, E.Y., 2018. Occurrence and fate of microplastics in wastewater treatment plants. In: Zeng, Eddy Y. (Ed.), Microplastic Contamination in Aquatic Environments. Elsevier, pp. 317–338.
Pazos, R., Maiztegui, T., Colautti, D., Paracampo, A., Gómez, N., 2017. Microplastics in gut contents of coastal freshwater fish from Río de la Plata estuary. Mar. Pollut. Bull. 122, 85–90.
Pazos, R.S., Bauer, D.E., Gómez, N., 2018. Microplastics integrating the coastal planktonic community in the inner zone of the Río de la Plata estuary (South America). Environ. Pollut. 243, 134–142.
Pegado, T.de Souza E Silva, Schmid, K., Winemiller, K.O., Chelazzi, D., Cincinelli, A., Dei, L., Giarrizzo, T., 2018. First evidence of microplastic ingestion by fishes from the Amazon River estuary. Mar. Pollut. Bull. 133, 814–821.
Peng, G., Zhu, B., Yang, D., Su, L., Shi, H., Li, D., 2017. Microplastics in sediments of the Changjiang estuary, China. Environ. Pollut. 225, 283–290.
Pillar V., V., 2004. MULTIV: Multivariate Exploratory Analysis, Randomization Testing and Bootstrap Resampling. User's Guide v. 2.3.10, Dep. de Ecol., UFRGS, Porto Alegre, Brazil.
Possatto, F.E., Barletta, M., Costa, M.F., Ivar do Sul, J.A., Dantas, D.V., 2011. Plastic debris ingestion by marine catfish: an unexpected fisheries impact. Mar. Pollut. Bull. 62, 1098–1102.
Ramos, J., Barletta, M., Costa, M.F., 2012. Ingestion of nylon threads by gerreidae while using a tropical estuary as foraging grounds. Aquat. Biol. 17, 29–34.
Ribic, C.A., Sheavly, S.B., Rugg, D.J., Erdmann, E.S., 2012. Trends in marine debris along the US pacific coast and Hawai'i 1998–2007. Mar. Pollut. Bull. 64, 994–1004.
RIC, Standard Practice for Coding Plastic Manufactured Articles for Resin Identification, 2016. Standard Practice for Coding Plastic Manufactured Articles for Resin Identification. ASTM International. Retrieved 21 January 2016.
Rochman, C.M., Tahir, A., Williams, S.L., Baxa, D.V., Lam, R., Miller, J.T., Teh, F., Werorilangi, S., Teh, S.J., 2015. Anthropogenic debris in seafood: plastic debris and fibers from textiles in fish and bivalves sold for human consumption. Sci. Rep. 5, 14340. https://doi.org/10.1038/srep14340.
Schmidt, C., Krauth, T., Wagner, S., 2017. Export of plastic debris by rivers into the sea. Environ. Sci. Technol. 51, 12246–12253.
Silva-Cavalcanti, J.S., Silva, J.D., França, E.J., Araújo, M.C., Gusmão, F.P., 2017. Microplastics ingestion by a common tropical freshwater fishing resource. Environ. Pollut. 221, 218–226.
Song, Y.K., Hong, S.H., Jang, M., Han, G.M., Rani, M., Lee, J., Shim, W.J., 2015. A comparison of microscopic and spectroscopic identification methods for analysis of microplastics in environmental samples. Mar. Pollut. Bull. 93, 202–209.
Su, L., Xue, Y., Li, L., Yang, D., Kolandhasamy, P., Li, D., Shi, H., 2016. Microplastics in taihu lake, China. Environ. Pollut. https://doi.org/10.1016/j.envpol.2016.06.036.
Sujathan, S., Kniggendorf, A.-K., Kumar, A., Roth, B., Rosenwinkel, K.-H., Nogueira, R., 2017. Heat and bleach: a cost-efficient method for extracting microplastics from return activated sludge. Arch. Environ. Contam. Toxicol. 73, 641–648.
The United Nations Environment Programme (UNEP), 2018. Report on the Status of Styrofoam and Plastic Bag Bans in the Wider Caribbean Region UNEP (DEPI)/CAR WG.39/INF.8.
United Nations Human Settlements Programme, 2016. Urbanization and Development: Emerging Futures, Nairobi, Kenya, p. 247. ISBN 978-92-1-133395-4.
van Wijnen J., J., Ragas, A., Kroeze, C., 2019. Modelling global river export of microplastics to the marine environment: Sources and future trends. Sci. Total Environ. 673, 392–401.
Vermeiren, P., Muñoz, C.C., Ikejima, K., 2016. Sources and sinks of plastic debris in estuaries: a conceptual model integrating biological, physical and chemical distribution mechanisms. Mar. Pollut. Bull. 113, 7–16.
Wang, J., Liu, X.D., Lu, J., 2012. urban river pollution control and remediation. Procedia Environ. Sci. 13, 1856–1862.
Wang, W., Ndungu, A.W., Li, Z., Wang, J., 2016. Microplastics pollution in inland freshwaters of China: a case study in urban surface waters of Wuhan, China. Sci. Total Environ. https://doi.org/10.1016/j. scitotenv.2016.09.213.
Wantzen, K.M., Alves, C.B.M., Badiane, S.D., Bala, R., Blettler, M., Callisto, M., Cao, Y., Kolb, M., Kondolf, G.M., Leite, M.F., Macedo, D.M., Mahdi, O., Neves, M., Peralta, M.E., Rotgé, V., Rueda-Delgado, G., Scharager, A., Serra-Llobet, A., Yengué, J.L., Zingraff-Hamed, A., 2019. Urban Stream and Wetland Restoration in the Global South—A DPSIR Analysis. Sustainability (in press).
Weinstein, J.E., Crocker, B.K., Gray, A.D., 2016. From macroplastic to microplastic: degradation of high-density polyethylene, polypropylene, and polystyrene in a salt marsh habitat. Environ. Toxicol. Chem. 35, 1632–1640.
Wong, C.M., Williams, C.E., Pittock, J., Collier, U., Schelle, P., 2007. World's Top 10 Rivers at Risk. World Wildlife Fund, Gland.
Xiong, X., Zhang, K., Chen, X., Shi, H., Luo, Z., Wu, C., 2018. Sources and distribution of microplastics in China's largest inland lake e Qinghai Lake. Environ. Pollut. 235, 899–906.
Xu, Z., Xu, J., Yin, H., Jin, W., Li, H., He, Z., 2019. Urban river pollution control in developing countries. Nat. Sustain. 2, 158–160.
Yonkos, L.T., Friedel, E.A., Perez-Reyes, A.C., Ghosal, S., Arthur, C., 2014. Microplastics in four estuarine rivers in the Chesapeake Bay, U.S.A. Environ. Sci. Technol. 48, 14195–14202.
Zbyszewski, M., Corcoran, P.L., Hockin, A., 2014. Comparison of the distribution and

degradation of plastic debris along shorelines of the Great Lakes, North America. J. Great Lakes Res. 40, 288–299.
Zhang, K., Gong, W., Lv, J., Xiong, X., Wu, C., 2015. Accumulation of floating microplastics behind the three Gorges dam. Environ. Pollut. 204, 117–123.
Zhang, K., Xiong, X., Hu, H., Wu, C., Bi, Y., Wu, Y., Zhou, B., Lam, P.K.S., Liu, J., 2017. Occurrence and characteristics of microplastic pollution in Xiangxi bay of three Gorges Reservoir, China. Environ. Sci. Technol. 51, 3794–3801.