



HAL
open science

Macroplastic and microplastic contamination assessment of a tropical river (Saigon River, Vietnam) transversed by a developing megacity.

Lisa Lahens, Emilie Strady, Thuy-Chung Kieu-Le, Rachid Dris, Kada Boukerma, Emmanuel Rinnert, Johnny Gasperi, Bruno Tassin

► To cite this version:

Lisa Lahens, Emilie Strady, Thuy-Chung Kieu-Le, Rachid Dris, Kada Boukerma, et al.. Macroplastic and microplastic contamination assessment of a tropical river (Saigon River, Vietnam) transversed by a developing megacity.. *Environmental Pollution*, 2018, pp.661-671. <10.1016/j.envpol.2018.02.005>. <hal-01711525>

HAL Id: hal-01711525

<https://enpc.hal.science/hal-01711525v1>

Submitted on 24 Jul 2025

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

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



Distributed under a Creative Commons CC BY 4.0 - Attribution - International License

Macroplastic and microplastic contamination assessment of a tropical river (Saigon River, Vietnam) transversed by a developing megacity

Lahens L¹, Strady E^{2,*}, Kieu-Le Tc³, Dris R⁴, Boukerma Kada⁵, Rinnert Emmanuel⁵, Gasperi J⁴, Tassin B⁴

¹ Université Paris-Est, LEESU, UMR MA 102 - AgroParisTech, 61 Avenue du Général de Gaulle, 94010, Créteil Cedex, France

² Univ. Grenoble Alpes, CNRS, IRD, Grenoble INP, IGE, F-38000, Grenoble, France

³ CARE, Ho Chi Minh City University of Technology, VNU-HCM, Viet Nam; Faculty of Geology and Petroleum Engineering, Ho Chi Minh City University of Technology, VNU-HCM, Viet Nam.

⁴ Université Paris-Est, LEESU, UMR MA 102 - AgroParisTech, 61 Avenue du Général de Gaulle, 94010, Créteil Cedex, France

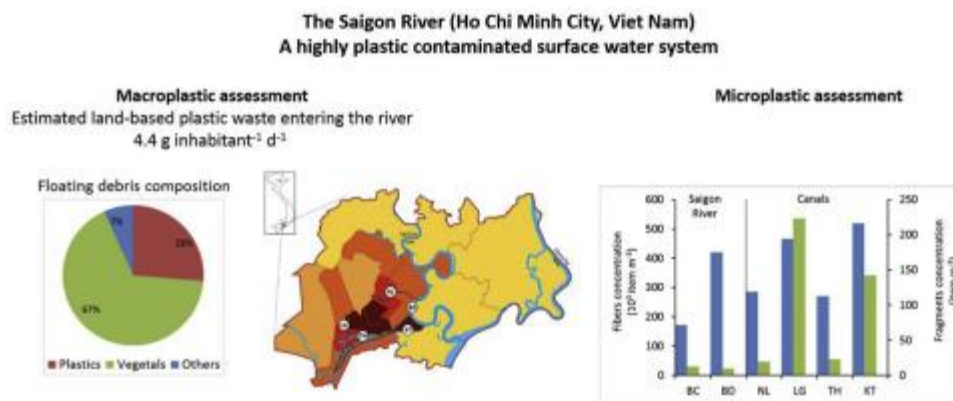
⁵ Ifremer, LDCM, Centre Bretagne, ZI de la Pointe du Diable, CS 10070, 29280, Plouzané, France.

* Corresponding author : Emilie Strady, email address : emilie.strady@ird.fr

Abstract :

Both macroplastic and microplastic contamination levels were assessed for the first time in a tropical river estuary system, i.e. the Saigon River, that traverses a developing South East Asian megacity, i.e. Ho Chi Minh City, Vietnam. The analysis of floating debris collected daily on the Nhieu Loc - Thi Nghe canal by the municipal waste management service shows that the plastic mass percentage represents 11-43%, and the land-based plastic debris entering the river was estimated from 0.96 to 19.91 g inhabitant⁻¹ d⁻¹, namely 350 to 7270 g inhabitant⁻¹ yr⁻¹. Microplastics were assessed in the Saigon River and in four urban canals by sampling bulk water for anthropogenic fiber analysis and 300 µm mesh size plankton net exposition for fragment analysis. Fibers and fragments are highly concentrated in this system, respectively 172,000 to 519,000 items m⁻³ and 10 to 223 items m⁻³. They were found in various colors and shapes with smallest size and surface classes being predominant. The macroplastics and fragments were mainly made of polyethylene and polypropylene while the anthropogenic fibers were mainly made of polyester. The relation between macroplastic and microplastic concentrations, waste management, population density and water treatment are further discussed.

Graphical abstract



Highlights

► Both macro and microplastic assessments were made in a tropical river canal system. ► About $4.4 \text{ g inhabitant}^{-1} \text{ d}^{-1}$ land-based plastic waste entering the river was estimated. ► High microplastic concentrations were found for both fibers and fragments. ► Characterization was carried out for macroplastics and microplastic.

Keywords : Anthropogenic fibers, Contamination, Macroplastic, Microplastic, Urban river

42

43 **1. Introduction**

44 Plastics are polymers designed to be lightweight, resistant, and durable (Thompson et
45 al., 2009). Their numerous advantages have led to a considerable increase in their production
46 in the past 60 years: 1.7 million metric tons produced in 1950 compared with 335 million metric
47 tons in 2017 worldwide (PlasticsEurope, 2018). However, plastic products are often
48 mismanaged or abandoned in illegal dumping sites (Barnes et al., 2009), which can lead to
49 environmental pollution. Their persistence in the environment makes them a good proxy of the
50 Anthropocene's exponential consumption (Steffen et al., 2007).

51 Plastics are estimated to represent between 50 and 80% of marine litter (Barnes et al.,
52 2009). Some chemicals are used as additives (e.g. alkylphenols, phthalates, bisphenol A,
53 organophosphates, PBDEs) during plastic production to give it particular properties (durability,
54 resistance, flexibility, reduced weight, ignition inhibition). On the one hand, plastics have the
55 capacity to adsorb persistent organic pollutants (POP) (Bakir et al., 2012, 2014) as well as trace
56 metals (Mato et al., 2001; Turner and Holmes, 2015) once they enter water bodies. On the other
57 hand, animals can become entangled in large items, or smaller plastic particles can be ingested
58 by fish and other organisms (Peters and Bratton, 2016; Sanchez et al., 2014; Silva-Cavalcanti
59 et al., 2017) and once they are ingested, they can clog the digestive system or release adsorbed
60 hazardous chemicals that could harm the biota (Bejgarn et al., 2015).

61 There has been increased interest since plastic was first assessed in the marine aquatic
62 environment in 1972 (Carpenter and Smith, 1972) and has been expanding to the continental
63 aquatic environment for the past few years (Dris et al., 2015a; Eerkes-Medrano et al., 2015;
64 Horton et al., 2017; Wagner et al., 2014). Plastic debris is generally assessed according to size:
65 macroplastics, i.e. plastic items superior to 5 mm, and microplastics, i.e. plastic items inferior
66 to 5 mm (Arthur et al., 2009; Barnes et al., 2009; Thompson et al., 2009). More precisely,

67 microplastics can be classified as (i) primary microplastics that are specifically engineered to
68 be used in this form, mainly in cosmetic products or as preproduction pellets, and as (ii)
69 secondary microplastics that come from the degradation of larger plastic items mainly due to
70 photo-degradation or mechanical action (Cooper and Corcoran, 2010; Derraik, 2002; Napper et
71 al., 2015; Williams and Simmons, 1996). In the continental aquatic environment, microplastic
72 assessment is conducted to estimate contamination of the environment and influence of
73 anthropogenic activities (e.g. Faure et al., 2015; Mani et al., 2015; Zhang et al., 2015), and also
74 to evaluate microplastic ingestion and impact on organisms (e.g. Jabeen et al., 2017; Jemec et
75 al., 2016; Rehse et al., 2016).

76 With regards to macroplastic, it is widely considered that around 80% of marine debris is from
77 land-based sources (Allsopp et al., 2006), even though a recent study estimated that 30% of
78 coastal plastic debris comes from marine activities and 47% corresponds to unidentifiable
79 fragments (Expéditions MED, 2016). Therefore, the macroplastic assessments from in situ
80 sampling aim to both quantify the floating debris (Morritt et al., 2014) and estimate the riverine
81 plastic fluxes or plastic exported to oceans (Estahbanati and Fahrenfeld, 2016; Gasperi et al.,
82 2014). The mass estimate of continental plastic waste entering the ocean can also be calculated
83 using a statistical approach based on governmental databases as exposed by Jambeck et al.
84 (2015). Indeed, the mass of mismanaged plastic waste generated by populations living within
85 50 km of the coast and the mass of plastic marine debris potentially entering the ocean were
86 estimated for each country. Over the ten top ranked countries, eight originate from Asia and
87 five are considered as lower middle-income countries by the World Bank: Indonesia,
88 Philippines, Vietnam, Sri Lanka and Bangladesh (Jambeck et al., 2015). It is important to note
89 that microplastic and macroplastic assessments are scarce in these lower middle-income
90 countries (Rochman et al., 2015) and that most studies are conducted in high-income countries
91 (e.g. Chae and An, 2017; Dris et al., 2015a; Eerkes-Medrano et al., 2015; Horton et al., 2017;

92 Wagner et al., 2014). Furthermore, Asia is one of the least studied continents in term of
93 microplastic contamination, which is contradictory with recent estimations showing that Asian
94 rivers introduce most of the worldwide continental plastic into the oceans (Lebreton et al.,
95 2017).

96 Ho Chi Minh City (HCMC) is the economic capital of Vietnam and is one of the most dynamic
97 developing megacities of South-East Asia. The increasing population (over 8 million
98 inhabitants, 3% growth per year; Kieu-Le et al., 2016) and its economic and industrial
99 development (7% of gross regional domestic product in the beginning of 2017, VGP News,
100 2017) have tremendous consequences on the quality of its aquatic environment, especially on
101 the Saigon River estuary system and its canals which intersect the city (Le et al., 2016; Strady
102 et al., 2017). Most of the city's wastewaters are discharged directly into the Saigon River and
103 its canals because only 10% of the total produced municipal wastewater is treated with activated
104 sludge (FAO, 2014). The major floating macroplastic contamination observed in the canals for
105 the last two decades (Givental, 2014) has led HCMC's environmental public institution to
106 reinforce the floating debris collection in the biggest canals. Between 2,000 to 13,000 metric
107 tons of floating debris are collected annually on the main urban canals (Kieu-Le et al., 2016).
108 Plastic waste mismanagement in Vietnam (4th world ranked according to Jambeck et al., 2015)
109 is a critical issue which has important environmental consequences. In this context, we selected
110 the Saigon River system to implement the first assessment of both macroplastic and
111 microplastic contamination in a tropical, urban river estuary system intersecting a South East
112 Asian megacity, HCMC, in the lower middle-income country Vietnam. The aims of the study
113 were (i) to evaluate the rate and type of macroplastics in the canal floating debris, (ii) to estimate
114 the land-based plastic debris entering the river per inhabitant, (iii) to assess the microplastic
115 (anthropogenic fibers and fragments) contamination levels in the canals and Saigon River
116 surface waters, and (iv) to compare the results to studies conducted in high-income countries.

117 For this comparison, we focus on the megacity of Paris crossed by the Seine River as a similar
118 methodology is used for both micro and macro plastics in each study (Dris et al., 2015b; Gasperi
119 et al., 2014).

120 **2. Materials and methods**

121 ***2.1 Study area***

122 The Saigon River is located in Southern Vietnam and is about 250 km long with a
123 catchment area of 4,717 km². It originates in southeastern Cambodia, flows through the
124 economic capital city of HCMC where the river is connected to a dense urban canal network
125 (700 km length), confluences with the Dong Nai River (470 km long), and then flows into the
126 Can Gio mangrove and South China Sea (Figure 1). The Saigon River is in a tropical monsoon
127 climate (one rainy and one dry season) and is affected by asymmetric semi-diurnal tides.
128 Despite a tidal range up to 3 meters, the resulting water discharge over a tidal cycle is low and
129 present a mean annual flow rate of 85 m³ s⁻¹ (Nguyen et al., 2011). The canal network dynamic
130 is partly controlled by floodgates and has low current velocity. The canal water quality is poor
131 (e.g. Strady et al., 2017) and has been deteriorating for the past four decades due to natural and
132 socioeconomic factors, including flood sediment deposition and maintenance neglect (Givental,
133 2014). The river catchment basin drains most of the untreated wastewater from dense urban
134 districts (up to 40,000 people km⁻², 2014, HCMC Statistical Office) and industrial zones (17
135 zones), which include textile, apparel, plastic and packaging production industries.

136 This study focusses on the Saigon River and four of its main canals (Figure 1). The macroplastic
137 assessment was conducted on the Nhieu Loc - Thi Nghe (NL) canal. We note that the floating
138 debris of this canal is collected daily by a municipal company (Kieu-Le et al., 2016). The
139 microplastic assessment was conducted on the Saigon River at Ben Cui (BC), 85 km upstream
140 HCMC, and at Bach Dang (BD), in the city center, and on the four main urban canals: Nhieu

141 Loc - Thi Nghe (NL), Tau Hu (TH), Kenh Te (KT) and Lo Gom (LG).

142 ***2.2 Macroplastic assessment***

143 ***Sampling***

144 Five representative sub-samples (3 to 5 kg each; 21 kg total) of floating debris were
145 collected on the NL canal in April 2016 (two campaigns). The floating debris collection (from
146 water surface to up down to 70 cm depth), under the supervision of a public company since
147 2012, aims mainly at improving the cleanness of the surface water for esthetic and population
148 perception purpose. Workers collect the floating debris manually from boats using nets (2 cm
149 mesh size) towed on each side of the boat and also by using landing nets when floating debris
150 is less accessible (Supporting Information Figure S1; Kieu-Le et al., 2016). We note that as the
151 mesh size is of 2 cm, the fraction of macroplastics between 5 mm and 2 cm is discarded and not
152 taken into account in the total mass calculation. The floating debris sub-samples were separated
153 in the laboratory into the following categories: vegetation, plastics and others. The plastics were
154 visually sorted into three groups: plastic bags, food packaging and other plastics. Finally, each
155 category and sub-category were weighed and the wet mass percentage of each was estimated.
156 The results are expressed in wet weight in order to be compatible with figures provided by the
157 company's database. A panel of plastics (41 plastic bags, 2 plastic cups, 2 straws and 2 plastic
158 spoons) was characterized by Fourier Transform InfraRed spectroscopy coupled to an
159 Attenuated Total Reflectance accessory (FTIR-ATR, UPEC, Créteil, France) using a database
160 containing spectra references of the most common polymers.

161 ***Land-based plastic release estimation calculation***

162 A rough estimation (minimum and maximum value) of land-based plastic waste weight
163 entering the river was calculated (in g inhabitant⁻¹ d⁻¹ and g inhabitant⁻¹ yr⁻¹) according to a set
164 of four assumptions. The first assumption is about the daily collected floating debris weight
165 which fluctuates daily between 10 and 12 metric tons d⁻¹ in accordance with the daily floating

166 debris collection weight database of the public collection company (Kieu-Le et al., 2016). The
167 second assumption refers to the percentage of floating debris captured by the collection team.
168 Although the floating debris collection on this canal seems to be visually very thorough, the
169 collection recovery efficiency is not quantified and varies daily according to the amount of
170 floating debris and the effectiveness of the worker. Accordingly, we estimated that the capture
171 efficiency varies between two extreme situations: an effective collection representing a 100%
172 recovery and a poor collection representing 50% recovery of the debris floating in the canal.
173 The third assumption relies on the percentage of plastic debris in the collected floating debris.
174 This percentage is estimated from the minimum, maximum and mean plastic percentage
175 weighed in the collected floating debris. The fourth assumption concerns the inhabitant
176 population estimations in the canal basin, which vary from 522,000 inhabitants (Thao and
177 Godfrey, 2014) to 1.2 million inhabitants (World Bank, 2012) depending on statistical sources.
178 We note here that as the floating debris are collected daily, we consider that the residence time
179 of the floating debris in the surface water is short enough to be excluded from the rough
180 estimation of land-based plastic waste weight entering the river.

181 ***2.3 Microplastic assessment***

182 ***Sampling***

183 Microplastics, composed of fragments and anthropogenic fibers, were sampled at two
184 sites of the Saigon River and on the four main canals during both rainy (December 2015) and
185 dry (April 2016) seasons from separate sampling. Contrary to previous study using an 80 μm
186 mesh size net to collect fibers (Dris et al 2015b), the high fibers contents measured in the river
187 water during a preliminary survey enabled us to consider a raw water sample for fibers analysis.
188 Thus at each sampling site, a sample of 300 mL bulk water was collected using a bucket for
189 fiber analysis. As commonly done in the literature (e.g. Eerkes-Medrano et al., 2015, Dris et
190 al., 2015a) fragments were collected using a 300 μm mesh size net exposed in surface water for

191 60 seconds, combined with a General Oceanic® flowmeter to determine the sampled water
192 volume. The net was then rinsed from the outside and the collected fraction was recovered in a
193 glass container.

194 ***Pre-treatment and density separation***

195 Both bulk and 300 µm fraction samples were treated based on a previous version of the
196 protocol of Mintenig et al. (2014). At first, 1 g of Sodium Dodecyl Sulfate (SDS, Merck®) was
197 added to the sample and kept in a closed bottle at 70°C for 24h in a laboratory oven. Then 1
198 mL of biozym SE (protease and amylase, Spinnrad®) and 1 mL of biozym F (lipase,
199 Spinnrad®) were added to the sample and kept at 40°C for 48h, before adding 15 mL of
200 hydrogen peroxide (H₂O₂, Merck®) and kept again at 40°C for 48h in the laboratory oven.
201 Then, a density separation step was performed using a saturated zinc chloride solution (ZnCl₂
202 Merck®, 1.6426 ± 0.0016 g cm⁻³). Lastly, the samples were filtered through glass fiber filters
203 (GF/D, 2.7 µm porosity) using a glassware filtration unit.

204 ***Sample analysis***

205 The filters were observed using a Leica MZ12 stereomicroscope at a 16 to 160-fold
206 magnification. We note that anthropogenic fibers and fragments were counted and measured
207 using the measuring tool of the image analysis software (Histolab®) with a lower length limit
208 set at 50 µm as smaller items were hard to identify. In this study, we defined as anthropogenic
209 fibers the synthetic fibers from petrochemical origin (e.g. polyester, polyamide,
210 polypropylene, etc), the artificial fibers from cellulose (e.g. viscose, rayon) and the natural fibers
211 (e.g. cotton, wool) used in textile and apparel industries. These fibers are transported in the
212 environment by water (Dris et al., 2018) and by air (Dris et al., 2016) and are ingested by
213 macrofauna (Remy et al., 2015). Anthropogenic fibers were exclusively counted and measured
214 (length in µm) on filters from bulk samples. Fragments were evaluated (length and surface in
215 µm and µm² respectively) from net sample filters. The thickness of fibers and fragments was

216 also measured on sub-samples by raising the particles with pliers. The composition of
217 anthropogenic fibers and fragments were evaluated by FTIR microscopy used in reflection and
218 transmission modes (IFREMER, Brest, France) on both fiber and fragment sub-samples: 76
219 fibers over a total of 725 fibers (about 10%) and 57 fragments over a total of 368 fragments
220 (about 15%). The FTIR microscopy measurements were performed with a Thermo IS50
221 infrared spectrometer coupled to an infrared Thermo Nicolet™ Continuum™ microscope with
222 a 15x IR objective and a Mercury Cadmium Telluride (MCT) single element detector cooled
223 with liquid nitrogen. The spectra resolution was 4 cm⁻¹. For fibers, each fiber without size
224 consideration was placed in a diamond micro-compression cell, one by one, and was flattened
225 between the two diamond anvils of a SPECAC® micro compression cell purchased from
226 Eurolabo (IFREMER, Brest, France). The identification of the fragment and anthropogenic
227 fiber' nature was performed using the Thermo IR polymer database.

228 *Expression of results*

229 The anthropogenic fiber and fragment observations were expressed in number of
230 particles per m³ (items m⁻³). We also estimated anthropogenic fiber and fragment particle mass
231 concentrations (mg m⁻³) based on two assumptions relative to polymer density and particle
232 volume. Concerning the estimation of polymer density, we took into account the lowest and
233 highest densities of polymers sampled in surface freshwaters (this study; Di and Wang et al.,
234 2017) in order to have a rough density estimation range: the lightest (0.9 g cm⁻³ – polypropylene
235 PP) and the heaviest (1.6 g cm⁻³ – polyvinyl chloride PVC). Concerning the estimation of
236 particle volume, the individual measured length of fibers and surface of fragments were taken
237 into account with a rough estimation of fibers diameter and fragments volume based on a
238 minima and maxima measurements made on fibers and fragments sub-samples, respectively
239 from a minimum of 8 μm to a maximum of 20 μm for fibers and a minimum of 10 μm and a
240 maximum of 40 μm for fragments.

241 **3. Results and discussion**

242 **3.1 Macroplastics**

243 In the Nhieu Loc - Thi Nghe (NL) canal, vegetation (e.g. water hyacinth and coconut)
244 made 67% of the wet weight while other debris (e.g. aluminum cans, glass bottles, clothes)
245 accounted for only 7% in mass. Plastic debris mass ranged from 12 to 43% of the total debris
246 weight with a mean of 26% (Figure 2.A). The sorting of plastic debris highlighted that plastic
247 bags represented 37% of the total plastic debris mass (Figure 2.B), namely 9% of the total debris
248 mass. These bags were found in many different colors and textures. Food packaging items made
249 of polystyrene represented 9 to 22% of the plastic mass (Figure 2.B) and 2 to 7% of the total
250 debris mass. Other plastic debris was composed of plastic bottles, drinking recipients and plastic
251 cutlery. This category constituted nearly half the weight of the plastic debris (Figure 2.B) and
252 around 11% of the total debris weight which is in the same range as observed in the Seine River
253 (Gasperi et al., 2014) and the Thames River, where it amounted to around 30% of plastic debris
254 (Morritt et al., 2014). The qualitative screening of a panel of plastic bags, cups, straws and
255 spoons showed that they were made of polyethylene (PE, 79%), polypropylene (PP, 15%) and
256 polyethylene terephthalate (PET, 4%) (Figure 2C). We note that one plastic bag was made of
257 both PE and PP (Figure 2.C). These proportions are coherent with the global production of
258 polymers (PlasticsEurope, 2016).

259 Considering the set of four assumptions used for the estimation of the land-based plastic release
260 to the river (section 2.2) and the estimated minimum and maximum plastic percentages (12 and
261 43%), the land-based plastic waste entering the NL canal was found to lie between 0.96 and
262 19.91 g inhabitant⁻¹ d⁻¹ (for calculation refers to Supporting Information Table S1), namely 350
263 to 7,270 g inhabitant⁻¹ yr⁻¹. The median was calculated at 4.43 g inhabitant⁻¹ d⁻¹, namely around
264 1,620 g inhabitant⁻¹ yr⁻¹. This daily land-based plastic waste entering the river per inhabitant
265 (0.96-19.91 g inhabitant⁻¹ d⁻¹) is smaller than the plastic release into the river (13.5-36.0 g

266 inhabitant⁻¹ d⁻¹) estimated by Jambeck et al. (2015) from governmental data using conversion
267 rates (e.g. 15 and 40%) of calculated mismanaged plastic waste into marine debris. This
268 difference of land-based plastic waste entering the river could be related to the difference of
269 methodology between the two studies. In the NL canal, only the floating debris superior to 2
270 cm is collected: the debris, including macroplastics, that sink or settle on the riverbed are
271 neglected. In the Thames River, Morritt et al. (2014) point out that a large, unseen volume of
272 submerged plastic is flowing into the marine environment. The underestimation of this
273 subsurface component in our study could lead to an underestimation of the floating debris and
274 therefore to a lower estimation of plastic release per inhabitant. Secondly, the estimation of the
275 daily land-based plastic waste entering the river per inhabitant of this study is partly based on
276 a field campaign conducted on the main canal of the country's economic capital, whereas the
277 release calculated by Jambeck et al. (2015) is based on a theoretical approach at the country
278 scale. It can be easily assumed that the waste management disparity observed at the scale of
279 two canals in HCMC by Kieu-Le et al. (2016) (i.e., technical support and constraint, human
280 resources), would be even more pronounced at the scale of the city or the country. A great
281 difference could be expected for instance between rural provinces and high-income provinces
282 such as HCMC. In a high-income country, the estimated plastic release to the river per
283 inhabitant of a similar river system, i.e. the Seine River crossing the megacity of Paris (10
284 million inhabitants) in France, is of 2.3 g inhabitant⁻¹ yr⁻¹ (e.g. 1.3-8.2 g inhabitant⁻¹ yr⁻¹, Gasperi
285 et al., 2014), namely 700 times lower than the plastic release calculated in HCMC's canal
286 (median: 1,620 g inhabitant⁻¹ yr⁻¹; min-max: 350- 7,270 g inhabitant⁻¹ yr⁻¹). Based on these
287 estimations, it appears that land-based plastic waste entering the river is directly related to waste
288 management at the national, regional or district level. On a global scale, plastic release found
289 in HCMC is one order of magnitude higher than world estimates of land-based plastic waste

290 entering the river per inhabitant (153 to 321 g inhabitant⁻¹ yr⁻¹) estimated by Lebreton et al.
291 (2017) from a worldwide modeling approach.

292 **3.2 Microplastics**

293 ***Anthropogenic fibers***

294 The anthropogenic fibers sampled in the four canals and in the Saigon River were found
295 in various colors and lengths (Figure 3). The fiber size distribution of each site was similar and
296 evidenced the predominance of small fibers (50-250 μm) (Table 1), the largest size classes (>
297 1,000 μm) being barely represented. The cumulative number of fibers per size class sampled in
298 the whole system (Figure 4) represents the distribution patterns observed in the Saigon River
299 canal system. The chemical characterization of 76 anthropogenic fibers selected randomly
300 showed (i) that 92% of the fibers were synthetic fibers, namely 70% of polyester (with 9% of
301 polyethylene terephthalate (PET)), 5% of polyethylene (PE) or polypropylene (PP), 4% of PP,
302 4% of PE-PP copolymer, 4% of rayon, 1% of PP-vistalon, 1% of viscose and 1% of acrylic, (ii)
303 that 5% of fibers were natural fibers namely 4% of cotton and 1% of wool and (iii) that 3% of
304 the fibers were unidentified. Surprisingly, in comparison to other studies assessing the
305 proportion of synthetic or anthropogenic fibers in continental aquatic environment (e.g. Dris et
306 al., 2018), the proportion of synthetic fibers in comparison to not synthetic fibers (e.g. viscose,
307 rayon, cotton, wool.) is extremely high. The proportion of synthetic fibers measured in the water
308 (i.e. 92%) is in accordance with the global fiber production market (95 million metric tons,
309 TextileExchange, 2017) which is dominated by synthetic fibers production (65 million metric
310 tons, TextileExchange, 2017) and especially polyester production (52 million metric tons
311 TextileExchange, 2017). It also could reflect the possible sources of anthropogenic fibers to the
312 aquatic environment from the textile and apparel industry in the surroundings of HCMC.

313 The anthropogenic fiber concentrations determined at each site ranged from 172,000 items m⁻³
314 ³ at BC to 519,000 items m⁻³ at KT (Figure 5), resulting in a variation of up to 3 folds between

315 sites. The corresponding estimated anthropogenic fiber mass concentrations ranged from 4.72
316 mg m^{-3} at BD considering the minimizing assumptions to 220.63 mg m^{-3} at KT considering the
317 maximizing assumptions, with an overall median at 31.71 mg m^{-3} (Supporting Information
318 Table S2). In the Saigon River, the difference of concentrations between the upstream site BC
319 (172,000 items m^{-3}), located in the countryside 85 km upstream of the city, and the city center
320 site BD (419,000 \pm 41,000 items m^{-3}) is lower than expected and difficult to explain as the
321 population densities between the two sites differ by more than one order of magnitude and many
322 textile and plastic industries are located up to 20 km upstream BD. We note that this unexpected
323 tendency was also observed in the Seine River between the sites located upstream and
324 downstream of the Paris megacity (Dris et al., 2015b). The concentrations in the canals varied
325 from 270,000 items m^{-3} in TH to 519,000 items m^{-3} in KT (Figure 5). The KT and LG canals
326 flow through the districts with the highest population densities ($> 40,000 \text{ km}^2$, Figure 1) with
327 many sub-canal tributaries. The absence of wastewater treatment plants (WWTP) and of
328 floating debris collection in this area lead respectively to a large amount of wastewater
329 discharge and to a visible littering in these canals. The lower concentrations observed in NL
330 and TH canals could be related to recent rehabilitation of these canals in terms of a drastic
331 reduction in illegal housing along the canals and the improvement of water quality carried out
332 by the World Bank (2012) and the Japan International Cooperation Agency improvement
333 program (JICA, 2001), respectively.

334 Compared to the anthropogenic fiber concentrations in other riverine systems, the fiber
335 concentrations measured in the Saigon River canal system are two order of magnitude higher
336 than in the Yangtze River (1911 \pm 701 items m^{-3} ; Wang et al., 2017) and Hanjiang River (2300
337 \pm 250 items m^{-3} ; Wang et al., 2017) in China, three to five orders of magnitude higher than in
338 the Seine River surface water (3-106 items m^{-3} ; Dris et al., 2015b; Dris et al., 2018) and six
339 orders of magnitude higher than the Swiss rivers Rhône, Aubonne, Venoge and Vuachère (0.7

340 items m^{-3} ; Faure et al., 2015). The predominance of small fibers (50-250 μm ; Table 1) observed
341 in the Saigon River canal system differed also from the distribution size observed in the Seine
342 River where the (250-450 μm) size is the dominant one (Dris et al., 2018). We note that despite
343 the difference of context between these three systems, the methodology used differed: fibers
344 were sampled in bulk water in the Saigon River, Yangtze River and in the Hanjiang River, using
345 a 80 μm mesh size net in the Seine River and a 300 μm mesh size net in the Swiss rivers. We
346 therefore conclude that the difference of sampling without net (bulk water) or with different
347 mesh size net is important and can introduce a bias as discussed by Dris et al. (2018) in the
348 determination of fiber concentrations and the dominant size class. Furthermore, the
349 contamination levels are comparable to the fiber concentrations measured in raw WWTP of the
350 Seine River basin (Dris et al., 2015b) and in influents of Dutch WWTP (Leslie et al., 2017),
351 and are even higher than fiber concentrations measured in the final WWTP effluents discharged
352 in the North Sea in Sweden (Magnusson and Norén, 2014), in the Baltic Sea in Finland (Talvitie
353 et al., 2015), and from several facilities in the USA (Mason et al., 2016). However, we have to
354 take into account for the site comparison that each study has a specific lower observation limit
355 for counting and measuring the fibers, from 10 to 300 μm , leading likely to a bias in
356 concentration estimation. Experimental studies have evidenced that washing machines can
357 discharge up to 728,000 fibers from a 6 kg wash load (Napper and Thompson, 2016), which
358 can lead to high fiber concentrations in effluent originating from washing machine discharge
359 (Browne et al., 2011; Dris et al., 2015b; Hartline et al., 2016; Napper and Thompson, 2016). In
360 Vietnam, the use of washing machines is less common than in high-income countries but we
361 do not exclude that both washing machines and handwashing could be sources of fiber
362 discharge in the canals and river. Moreover, the presence of many textile industries in HCMC
363 (as discussed above) coupled to the quasi absence of treatment of domestic (i.e. 10%; FAO,
364 2014) and industrial wastewaters (e.g. discharge into the river estimated at 200,000 $\text{m}^3 \text{d}^{-1}$; Vo,

2007) are an important source of fiber contamination in the Saigon River and canal system. Thus, compared to concentrations measured worldwide in various systems, the fiber concentrations measured in the Saigon River canal system might reflect the consequences of inadequate wastewater treatment on fiber concentrations in an aquatic system.

Fragments

In the urban canals and the Saigon River, a high variety of colors and shapes of fragments (i.e. irregular and regular shapes, lines, ovals, films and foams) were observed, with a predominance of the blue color and the irregular shape (Figure 3). No microbeads were observed. Similar microplastic fragment diversity and absence of microbeads were observed in the Yangtze River (Zhang et al., 2015) and the Three Gorges Reservoir (Zhang et al., 2017) in China, while the presence of microbeads were observed in the Rhine River, Germany (Mani et al., 2015). The absence of microbeads in the Saigon River system could be explained by a difference in consumer habits or cosmetics composition between countries, as microbeads mainly originate from facial cleansers (Napper et al., 2015). The chemical characterization of 57 fragments selected randomly showed that 40.4% of the fragments were made of PE, with 19.3% of oxidized PE, 24.6% of PP, 14% of PE/PP, 3.5% of PS, 1.8% of PA, 1.8% of PVC, 1.8% of polyepoxy, 1.8% of polyester, 1.8% of PE/ethyl acrylate, while 8.8% of fragments appeared to be non-plastic (cellulose, mineral). Considering that this characterization is representative of all the fragments measured (368 fragments in total), we evaluate that the visual determination of the plastic fragments on filters has a precision of 9%. The proportions of polymers are in agreement with global polymer production (PlasticsEurope, 2016) and with the characterization of macroplastics in the NL canal.

The cumulative number of fragments for all sites showed that the smallest fragment surface size class (500 to 50,000 μm^2) was predominant and represented half of the total number of fragments (Table 1; Figure 6). Contrary to fibers, the fragment number distribution per size

390 class varied between the sampling sites (Table 1; Figure 7). The smallest surface size class was
391 not systematically the dominant one. In TH, 70% of the fragments were in the smallest size
392 class whereas in BC, BD, and NL, the smallest fraction represented only 30% of fragments. In
393 terms of maximum length of the fragments, the mean value was around 835 μm and the median
394 around 520 μm .

395 The fragment concentrations varied from 10 items m^{-3} in the Saigon River at BC to 223 items
396 m^{-3} in the canals at LG (Figure 8). As observed for fibers, the difference of fragment
397 concentrations is unexpectedly low between BC and BD. In the canals, the highest fragment
398 concentrations were measured in LG and KT (223 items m^{-3} and 142 items m^{-3} respectively)
399 while they were 10 times lower in NL and TH (19 items m^{-3} and 23 items m^{-3} respectively).
400 This pattern is similar to fiber concentrations and could have the same explanation of lower
401 concentrations in NL and TH canals: the difference of population density and resulting inputs,
402 the effect of the canals rehabilitation, and the presence of floating debris collection on these
403 two canals (Kieu-Le et al., 2016) reducing macroplastic contamination and thus its degradation
404 into microplastic fragments. We also hypothesized that the water stagnation observed in LG
405 canal can contribute to microplastic accumulation, as observed in the Three Gorges Dam system
406 in China (Zhang et al., 2015). However, at KT, the water is not stagnant and is largely
407 influenced by tides. This canal is subjected to intense maritime traffic and boat habitation that
408 could import significant amounts of fragments from wastewater discharge and littering.

409 The fragment concentrations comparison to high-income countries emphasizes that higher
410 concentrations were found in the Saigon River system than in the Swiss rivers Rhône, Aubonne,
411 Venoge and Vuachère (Faure et al., 2015) and in the Tamar Estuary in the UK (Sadri and
412 Thompson, 2014), and the same range of concentrations was found in the Amsterdam canals in
413 the Netherlands in bulk water samples with an observation limit of 10 μm (Leslie et al., 2017).
414 Interestingly, the fragment concentrations in the river and canals are in the same range as in the

415 Yangtze, Jiaojiang, Oujiang and Minjiang estuaries in China (e.g. the first ranked country for
416 mismanaged plastic waste and marine debris entering the ocean; Jambeck et al., 2015) despite
417 that they have an observation limit of 500 μm (Zhao et al., 2014, 2015) compared to the 50 μm
418 limit in the present study. We also note here that the estimated concentrations of fragments
419 depend on the observation limit, i.e. a lowest limit providing a higher concentration, the smallest
420 fractions being the most abundant (this study, Wang et al., 2017).

421 The fragment mass concentrations were estimated for each site from 0.01 mg m^{-3} in TH with
422 the minimizing assumptions to 1.91 mg m^{-3} in LG with the maximizing assumptions, with an
423 overall median at 0.15 mg m^{-3} (for calculation see Supporting Information Table S3). The
424 highest concentrations were found in KT and LG. Although the numerical concentration of
425 fragments in TH was higher than in the Saigon River and NL, its mass concentration was
426 smaller which can be explained by the scarcity of large fragments (Figure 7) as mass estimations
427 were done based on surface measurements.

428 *Anthropogenic fibers versus fragments*

429 Globally, the concentrations of anthropogenic fibers were greater than those of
430 fragments by 3-4 orders of magnitude (fibers: 172,000 to 519,000 items m^{-3} ; fragments: 10 to
431 223 items m^{-3}). The mass concentrations were 1-3 orders of magnitude higher for fibers than
432 for fragments (anthropogenic fibers: 4.72 to 220.63 mg m^{-3} ; fragments: 0.01 to 1.91 mg m^{-3}).
433 The greater surface and associated calculated volume of fragments may lead to a higher mass
434 and consequently reduce the orders of magnitude from numerical to mass concentrations.
435 Considering the mass estimation assumptions, their robustness can be questioned as mass
436 concentrations can have a 2-fold variation depending on the assumption on fragments thickness
437 and by one order of magnitude when fiber diameter is changed. Nonetheless, the assumptions
438 regarding plastic density are based on actual polymer densities and even with variations of these
439 densities, mass concentrations stay in the same range. In this river canal system, the total

440 suspended matter concentrations are in the range of 30-140 mg L⁻¹(Strady et al., 2017). It is
441 thus interesting to state that anthropogenic fiber mass concentrations and fragment mass
442 concentrations represent roughly one per thousand and one per million of TSS mass
443 concentrations, respectively.

444 **4. Conclusions**

445 The study of the tropical Saigon River canal system, crossing a developing South East
446 Asian megacity, has highlighted high macroplastic and microplastic contamination. Land-based
447 macroplastics entering the river, estimated between 0.96 and 19.91 g inhabitant⁻¹ d⁻¹, seems to
448 be related to local habits and waste management. The high concentrations of microplastics
449 (fragments and anthropogenic fibers) in surface waters is related to the presence of several
450 textile and plastics industries in the vicinity of HCMC as well as high macroplastic
451 contamination and paucity of wastewater treatment in Vietnam. This data can thus serve waste
452 management regulation agencies with their implementation of regulations on plastic litter and
453 collection. This study, combining both macroplastic and microplastic assessments, has
454 evidenced the close relationship between these two contaminants and the impact of
455 macroplastic content on microplastic contamination via the assumed fragmentation and
456 degradation throughout the size spectrum continuum. We therefore recommend that the link
457 between these two size classes should not be overlooked in further studies. This study has also
458 been the first to point out differences of microplastic contamination in the continental aquatic
459 environments of high-income countries and lower middle-income countries. As plastic
460 production is likely to increase over the next few years, macro- and microplastic contamination
461 will remain an important concern: a better understanding of macro- and microplastic pollution,
462 sources, variability and degradation processes is thus imperative.

463 **Acknowledgments**

464 This study has been conducted under the framework of CARE-RESCIF initiative along
465 with the Urban Development Management Support Centre of HCMC (PADDI) and the
466 Department of Natural Resources and Environment of HCMC (DONRE). This study was
467 funded by incentive grants from the French Research Institute of Sustainable Development
468 (IRD). The PhD of Rachid Dris was funded by the region Île-de-France Research Network on
469 Sustainable Development (R2DS Ile-de-France, Grant N° 2013-2). The authors thank Phan Thi
470 San Ha, Joanne Aimé, Nguyen Truong An (CARE), Fanny Quertamp and Morgane Perset
471 (PADDI) for their local support and help during sampling campaigns, and Anne Chabas (LISA,
472 Laboratoire Interuniversitaire des Systèmes Atmosphériques) for technical support. The authors
473 thank also the anonymous reviewers for their comments which improve the quality of the
474 manuscript.

475 **References**

- 476 Allsopp, M., Walters, A., Santillo, D., and Johnston, P. (2006). Plastic debris in the world's
477 oceans.
- 478 Arthur, C., Baker, J., and Bamford, H. (2009). Proceedings of the International Research
479 Workshop on the Occurrence, Effects, and Fate of Microplastic Marine Debris, September 9-
480 11, 2008.
- 481 Bakir, A., Rowland, S.J., and Thompson, R.C. (2012). Competitive sorption of persistent
482 organic pollutants onto microplastics in the marine environment. *Mar. Pollut. Bull.* *64*, 2782–
483 2789.
- 484 Bakir, A., Rowland, S.J., and Thompson, R.C. (2014). Enhanced desorption of persistent
485 organic pollutants from microplastics under simulated physiological conditions. *Environ.*
486 *Pollut.* *185*, 16–23.
- 487 Barnes, D.K.A., Galgani, F., Thompson, R.C., and Barlaz, M. (2009). Accumulation and
488 fragmentation of plastic debris in global environments. *Philos. Trans. R. Soc. B Biol. Sci.*
489 *364*, 1985–1998.
- 490 Bejgarn, S., MacLeod, M., Bogdal, C., and Breitholtz, M. (2015). Toxicity of leachate from
491 weathering plastics: An exploratory screening study with *Nitocra spinipes*. *Chemosphere* *132*,
492 114–119.

493 Browne, M.A., Crump, P., Niven, S.J., Teuten, E., Tonkin, A., Galloway, T., and Thompson,
494 R. (2011). Accumulation of Microplastic on Shorelines Woldwide: Sources and Sinks.
495 *Environ. Sci. Technol.* *45*, 9175–9179.

496 Carpenter, E.J., and Smith, K.L. (1972). Plastics on the Sargasso sea surface. *Science* *175*,
497 1240–1241.

498 Chae, Y., and An, Y.-J. (2017). Effects of micro- and nanoplastics on aquatic ecosystems:
499 Current research trends and perspectives. *Mar. Pollut. Bull.*

500 Cooper, D.A., and Corcoran, P.L. (2010). Effects of mechanical and chemical processes on
501 the degradation of plastic beach debris on the island of Kauai, Hawaii. *Mar. Pollut. Bull.* *60*,
502 650–654.

503 Derraik, J.G.. (2002). The pollution of the marine environment by plastic debris: a review.
504 *Mar. Pollut. Bull.* *44*, 842–852.

505 Di, W., and Wang, J., (2017). Microplastics in surface waters and sediments of the Three
506 Gorges Reservoir, China. *Sci. Tot. Environ.* <https://doi.org/10.1016/j.scitotenv.2017.10.150>

507 Dris, R., Imhof, H., Sanchez, W., Gasperi, J., Galgani, F., Tassin, B., and Laforsch, C.
508 (2015a). Beyond the ocean: Contamination of freshwater ecosystems with (micro-) plastic
509 particles. *Environ. Chem.* *32*.

510 Dris, R., Gasperi, J., Rocher, V., Saad, M., Renault, N., and Tassin, B. (2015b). Microplastic
511 contamination in an urban area: a case study in Greater Paris. *Environ. Chem.*,pp2015.

512 Dris, R., Gasperi, J., Saad, M., Mirande, C., Tassin, B., 2016. Synthetic fibers in atmospheric
513 fallout: a source of microplastics in the environment? *Mar. Pollut.Bull.* *104*, 290-293.
514

515 Dris, R., Gasperi, J., Rocher V., Tassin, B., 2018. Synthetic and non-synthetic anthropogenic
516 fibers in a river under the impact of Paris Megacity: Sampling methodological aspects and
517 flux estimations. *Sc. Tot. Environ.*, *618*, 157–164.
518

519 Eerkes-Medrano, D., Thompson, R.C., and Aldridge, D.C. (2015). Microplastics in freshwater
520 systems: A review of the emerging threats, identification of knowledge gaps and prioritisation
521 of research needs. *Water Res.* *75*, 63–82.

522 Estahbanati, S., and Fahrenfeld, N.L. (2016). Influence of wastewater treatment plant
523 discharges on microplastic concentrations in surface water. *Chemosphere* *162*, 277–284.

524 Expéditions MED (2016). Opération déchets côtiers.

525 FAO (2014). Aquastat: global information system on water and agriculture.

526 Faure, F., Demars, C., Wieser, O., Kunz, M., and De Alencastro, L.F. (2015). Plastic pollution
527 in Swiss surface waters: Nature and concentrations, interaction with pollutants. *Environ.*
528 *Chem.* *12*, 582–591.

529 Gasperi, J., Dris, R., Bonin, T., Rocher, V., and Tassin, B. (2014). Assessment of floating
530 plastic debris in surface water along the Seine River. *Environ. Pollut.* *195*, 163–166.

- 531 Givental, E. (2014). The Ho Chi Minh City canals: assessing vulnerability and resilience
532 factors. *Yearb. Assoc. Pac. Coast Geogr.* 76, 49–67.
- 533 Hartline, N.L., Bruce, N.J., Karba, S.N., Ruff, E.O., Sonar, S.U., and Holden, P.A. (2016).
534 Microfiber Masses Recovered from Conventional Machine Washing of New or Aged
535 Garments. *Environ. Sci. Technol.* 50, 11532–11538.
- 536 Horton, A.A., Walton, A., Spurgeon, D.J., Lahive, E., and Svendsen, C. (2017). Microplastics
537 in freshwater and terrestrial environments: Evaluating the current understanding to identify
538 the knowledge gaps and future research priorities. *Sci. Total Environ.* 586, 127–141.
- 539 Jabeen, K., Su, L., Li, J., Yang, D., Tong, C., Mu, J., and Shi, H. (2017). Microplastics and
540 mesoplastics in fish from coastal and fresh waters of China. *Environ. Pollut.* 221, 141–149.
- 541 Jambeck, J.R., Geyer, R., Wilcox, C., Siegler, T.R., Perryman, M., Andrady, A., Narayan, R.,
542 and Law, K.L. (2015). Plastic waste inputs from land into the ocean. *Science* 347, 768–771.
- 543 Jemec, A., Horvat, P., Kunej, U., Bele, M., and Kržan, A. (2016). Uptake and effects of
544 microplastic textile fibers on freshwater crustacean *Daphnia magna*. *Environ. Pollut.* 219,
545 201–209.
- 546 JICA (2001). The Detailed Design Study on HCMC Water Environment Improvement Project
547 - Chapter 3 Tau Hu - Ben Nghe canal improvement.
- 548 Kieu-Le, T.-C., Strady, E., and Perset, M. (2016). Life cycle of floating debris in the canals of
549 Ho Chi Minh City (PADDI).
- 550 Le, T.M.T., Nguyen Phuoc, D., Dinh Quoc, T., Ngo, H.H., and Do Hong Lan, C. (2016).
551 Presence of e-EDCs in surface water and effluents of pollution sources in Sai Gon and Dong
552 Nai river basin. *Sustain. Environ. Res.* 26, 20–27.
- 553 Lebreton, L.C.M., van der Zwet, J., Damsteeg, J.-W., Slat, B., Andrady, A., and Reisser, J.
554 (2017). River plastic emissions to the world's oceans. *Nat. Commun.* 8, 15611.
- 555 Leslie, H.A., Brandsma, S.H., van Velzen, M.J.M., and Vethaak, A.D. (2017). Microplastics
556 en route: Field measurements in the Dutch river delta and Amsterdam canals, wastewater
557 treatment plants, North Sea sediments and biota. *Environ. Int.* 101, 133–142.
- 558 Löder, M. and Gerdts, G., Methodology Used for the Detection and Identification of
559 Microplastics—A Critical Appraisal, 2015. In: *Marine Anthropogenic Litter*, M. Bergmann, L.
560 Gutow and M. Klages (editors) *Marine Anthropogenic Litter*, Berlin, Springer, 447 p.,
561 ISBN: [978-3-319-16510-3](https://doi.org/10.1007/978-3-319-16510-3) . doi: [10.1007/978-3-319-16510-3_8](https://doi.org/10.1007/978-3-319-16510-3_8)
- 562
- 563
- 564 Magnusson, K., and Norén, F. (2014). Screening of microplastic particles in and down-stream
565 a wastewater treatment plant (Swedish Environmental Research Institute: Stockholm,
566 Sweden).
- 567 Mani, T., Hauk, A., Walter, U., and Burkhardt-Holm, P. (2015). Microplastics profile along
568 the Rhine River. *Sci. Rep.* 5, 17988.

569 Mason, S.A., Garneau, D., Sutton, R., Chu, Y., Ehmann, K., Barnes, J., Fink, P.,
570 Papazissimos, D., and Rogers, D.L. (2016). Microplastic pollution is widely detected in US
571 municipal wastewater treatment plant effluent. *Environ. Pollut.* *218*, 1045–1054.

572 Mato, Y., Isobe, T., Takada, H., Kanehiro, H., Ohtake, C., and Kaminuma, T. (2001). Plastic
573 resin pellets as a transport medium for toxic chemicals in the marine environment. *Environ.*
574 *Sci. Technol.* *35*, 318–324.

575 Mintenig, S., Int-Veen, I., Löder, M., and Gerdt, G. (2014). Mikroplastik in ausgewählten
576 Kläranlagen des Oldenburgisch - Ostfriesischen Wasserverbandes (OOWV) in Niedersachsen
577 (Alfred-Wegener-Institut, Helmholtz-Zentrum für Polar-und Meeresforschung).

578 Morritt, D., Stefanoudis, P.V., Pearce, D., Crimmen, O.A., and Clark, P.F. (2014). Plastic in
579 the Thames: A river runs through it. *Mar. Pollut. Bull.* *78*, 196–200.

580 Napper, I.E., and Thompson, R.C. (2016). Release of synthetic microplastic plastic fibres
581 from domestic washing machines: Effects of fabric type and washing conditions. *Mar. Pollut.*
582 *Bull.* *112*, 39–45.

583 Napper, I.E., Bakir, A., Rowland, S.J., and Thompson, R.C. (2015). Characterisation, quantity
584 and sorptive properties of microplastics extracted from cosmetics. *Mar. Pollut. Bull.* *99*, 178–
585 185.

586 Nguyen, T.V.H., Takizawa, S., Oguma, K., and Nguyen, V.P. (2011). Sources and leaching of
587 manganese and iron in the Saigon River Basin, Vietnam. *Water Sci. Technol.* *63*, 2231.

588 Peters, C.A., and Bratton, S.P. (2016). Urbanization is a major influence on microplastic
589 ingestion by sunfish in the Brazos River Basin, Central Texas, USA. *Environ. Pollut.* *210*,
590 380–387.

591 PlasticsEurope (2018). *Plastics - the Facts 2017*, An analysis of European latest plastics
592 production, demand and waste data. *Plast. Eur. Assoc. Plast. Manuf. Bruss.*

593 Rehse, S., Kloas, W., and Zarfl, C. (2016). Short-term exposure with high concentrations of
594 pristine microplastic particles leads to immobilisation of *Daphnia magna*. *Chemosphere* *153*,
595 91–99.

596 Remy F., Collard F., Gilbert B., Compère P., Eppe G., Lepoint G. 2018. When Microplastic Is
597 Not Plastic: The Ingestion of Artificial Cellulose Fibers by Macrofauna Living in Seagrass
598 Macrophytodetritus. *Environ. Sci. Technol.*, 2015, 49, pp 11158–11166.
599

600 Rochman, C.M., Tahir, A., Williams, S.L., Baxa, D.V., Lam, R., Miller, J.T., Teh, F.-C.,
601 Werorilangi, S., and Teh, S.J. (2015). Anthropogenic debris in seafood: Plastic debris and
602 fibers from textiles in fish and bivalves sold for human consumption. *Sci. Rep.* *5*.

603 Sadri, S.S., and Thompson, R.C. (2014). On the quantity and composition of floating plastic
604 debris entering and leaving the Tamar Estuary, Southwest England. *Mar. Pollut. Bull.* *81*, 55–
605 60.

606 Sanchez, W., Bender, C., and Porcher, J.-M. (2014). Wild gudgeons (*Gobio gobio*) from
607 French rivers are contaminated by microplastics: Preliminary study and first evidence.
608 *Environ. Res.* *128*, 98–100.

609 Silva-Cavalcanti, J.S., Silva, J.D.B., França, E.J. de, Araújo, M.C.B. de, and Gusmão, F.
610 (2017). Microplastics ingestion by a common tropical freshwater fishing resource. *Environ.*
611 *Pollut.* *221*, 218–226.

612 Steffen, W., Crutzen, P.J., and McNeill, J.R. (2007). The Anthropocene: Are Humans Now
613 Overwhelming the Great Forces of Nature. *AMBIO J. Hum. Environ.* *36*, 614–621.

614 Strady, E., Dang, V.B.H., Némery, J., Guédron, S., Dinh, Q.T., Denis, H., and Nguyen, P.D.
615 (2017). Baseline seasonal investigation of nutrients and trace metals in surface waters and
616 sediments along the Saigon River basin impacted by the megacity of Ho Chi Minh (Vietnam).
617 *Environ. Sci. Pollut. Res.* *24*, 3226–3243.

618 Talvitie, J., Heinonen, M., Paakkonen, J.-P., Vahtera, E., Mikola, A., Setälä, O., and Vahala,
619 R. (2015). Do wastewater treatment plants act as a potential point source of microplastics?
620 Preliminary study in the coastal Gulf of Finland, Baltic Sea. *Water Sci. Technol.* *72*, 1495–
621 1504.

622 Textile Exchange, 2017. Preferred Fiber and material market report 2017.
623 <http://textileexchange.org/downloads/2017-preferred-fiber-materials-market-report/>
624
625

626 Thao, V., and Godfrey, C. (2014). The resurrection of Nhieu Loc-Thi Nghe: How HCMC
627 brought the black water canal back from the dead. *Thanh Nien Dly.*

628 Thompson, R.C., Moore, C.J., vom Saal, F.S., and Swan, S.H. (2009). Plastics, the
629 environment and human health: current consensus and future trends. *Philos. Trans. R. Soc. B*
630 *Biol. Sci.* *364*, 2153–2166.

631 Turner, A., and Holmes, L.A. (2015). Adsorption of trace metals by microplastic pellets in
632 fresh water. *Environ. Chem.* *12*, 600–610.

633 VGP News (2017). Municipal Department of Planning and Investment, the online newspaper
634 of the government of the Socialist Republic of Viet Nam. <http://news.chinhphu.vn/>

635 Vo, P.L. (2007). Urbanization and water management in Ho Chi Minh City, Vietnam-issues,
636 challenges and perspectives. *GeoJournal* *70*, 75–89.

637 Wagner, M., Scherer, C., Alvarez-Muñoz, D., Brennholt, N., Bourrain, X., Buchinger, S.,
638 Fries, E., Grosbois, C., Klasmeier, J., Marti, T., et al. (2014). Microplastics in freshwater
639 ecosystems: what we know and what we need to know. *Environ. Sci. Eur.* *26*, 1–9.

640 Wang, W., Ndungu, A.W., Li, Z., Wang, J., 2017. Microplastics pollution in inland
641 freshwaters of China: A case study in urban surface waters of Wuhan, China. *Sc. Tot.*
642 *Environ.* *575*, 1369-1374.
643

644 Williams, A.T., and Simmons, S.L. (1996). The degradation of plastic litter in rivers:
645 implications for beaches. *J. Coast. Conserv.* *2*, 63–72.

646 World Bank (2012). Vietnam - Ho Chi Minh City Environmental Sanitation (Nhieu Loc-Thi
647 Nghe Basin) Project (The World Bank).

- 648 Zhang, K., Gong, W., Lv, J., Xiong, X., and Wu, C. (2015). Accumulation of floating
649 microplastics behind the Three Gorges Dam. *Environ. Pollut.* *204*, 117–123.
- 650 Zhang, K., Xiong, X., Hu, H., Wu, C., Bi, Y., Wu, Y., Zhou, B., Lam, P.K.S., and Liu, J.
651 (2017). Occurrence and Characteristics of Microplastic Pollution in Xiangxi Bay of Three
652 Gorges Reservoir, China. *Environ. Sci. Technol.* *51*, 3794–3801.
- 653 Zhao, S., Zhu, L., Wang, T., and Li, D. (2014). Suspended microplastics in the surface water
654 of the Yangtze Estuary System, China: First observations on occurrence, distribution. *Mar.*
655 *Pollut. Bull.* *86*, 562–568.
- 656 Zhao, S., Zhu, L., and Li, D. (2015). Microplastic in three urban estuaries, China. *Environ.*
657 *Pollut.* *206*, 597–604.
- 658

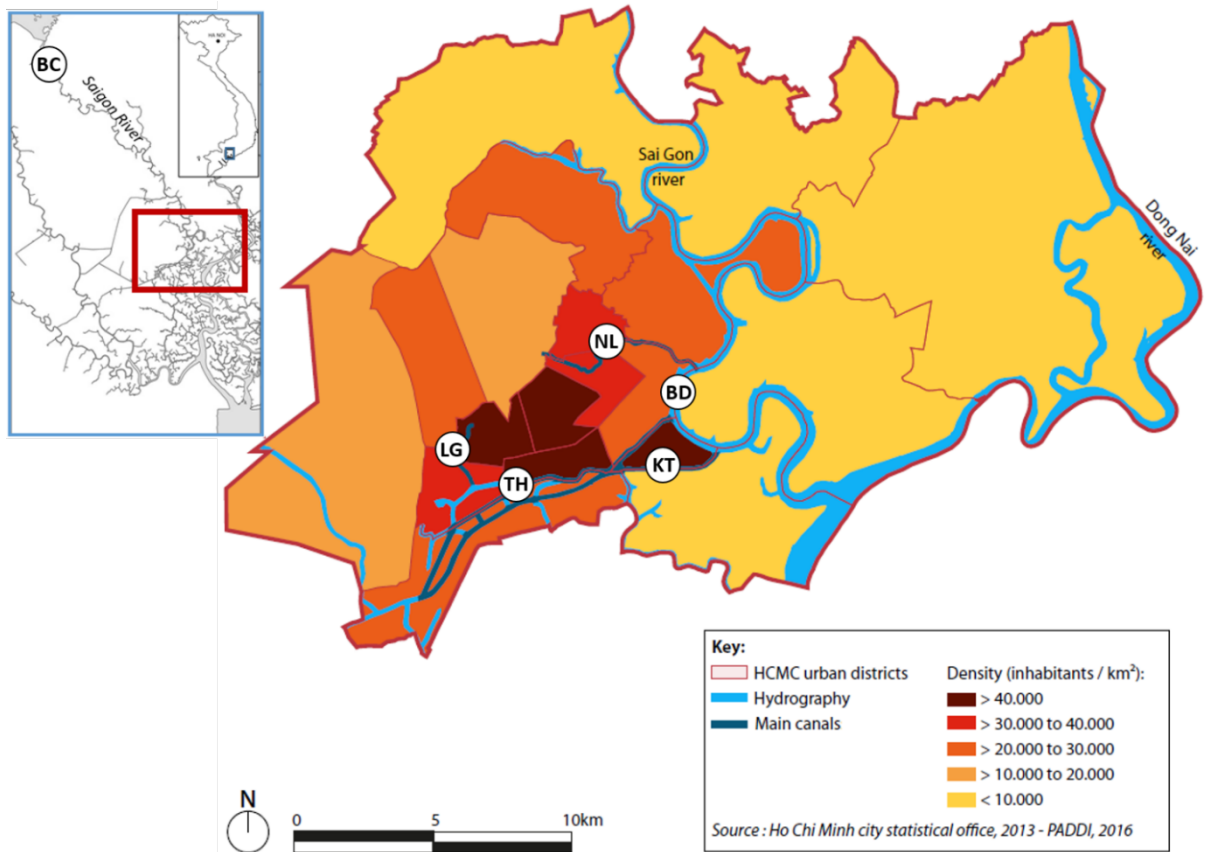


Figure 1: Map of the sampling sites. BC: Ben Cui, BD: Bach Dang, NL: Nhieu Loc - Thi Nghe, KT: Kenh Te, TH: Tau Hu, LG: Lo Gom. (adapted from Kieu-Le et al., 2016).

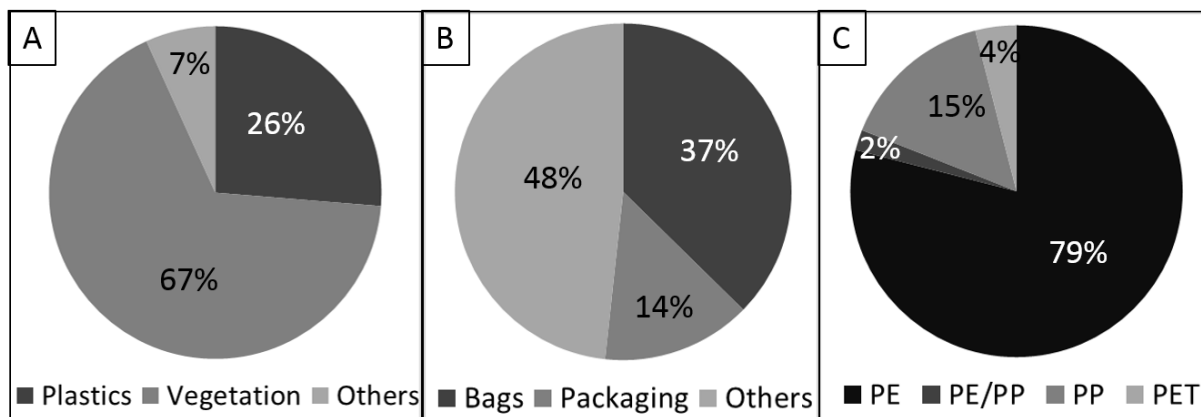


Figure 2: Macroplastic floating debris sort out and characterization at Nhieu Loc - Thi Nghe: A. Total floating debris sort out: plastic items, vegetation (water hyacinth, coconut), and others (shoes, mattress, wood furniture), B: Plastic debris sort out: plastic bags, plastic packaging (food take-out boxes, containers), and others (plastic bottles, food wrappings, cups, cutlery), C: Type of plastics for bBags and other plastics characterization: polypropylene (PP), polyethylene (PE), mix of PE and PP, and polyethylene terephthalate (PET).

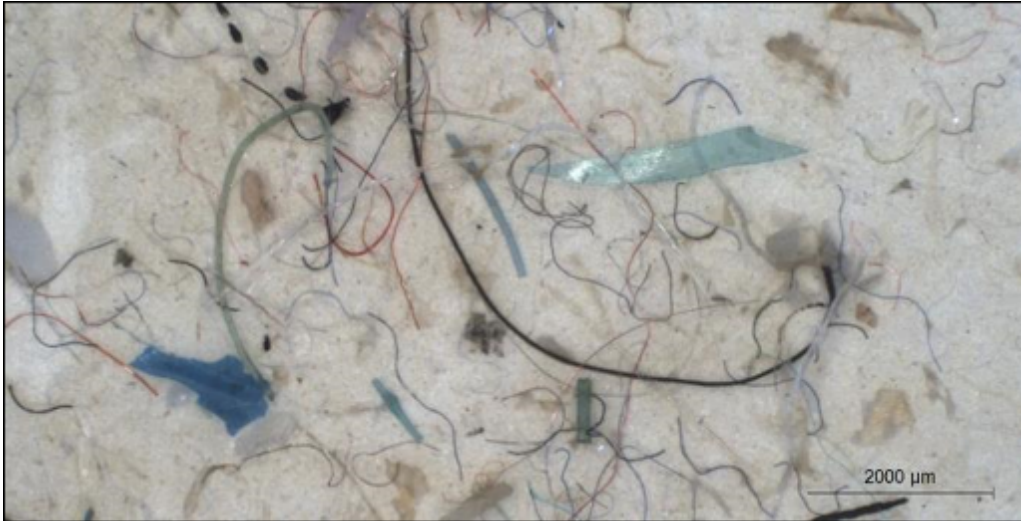


Figure 3: Stereomicroscope image of fibers and fragments collected in the Saigon River canal system on a GF/D filter.

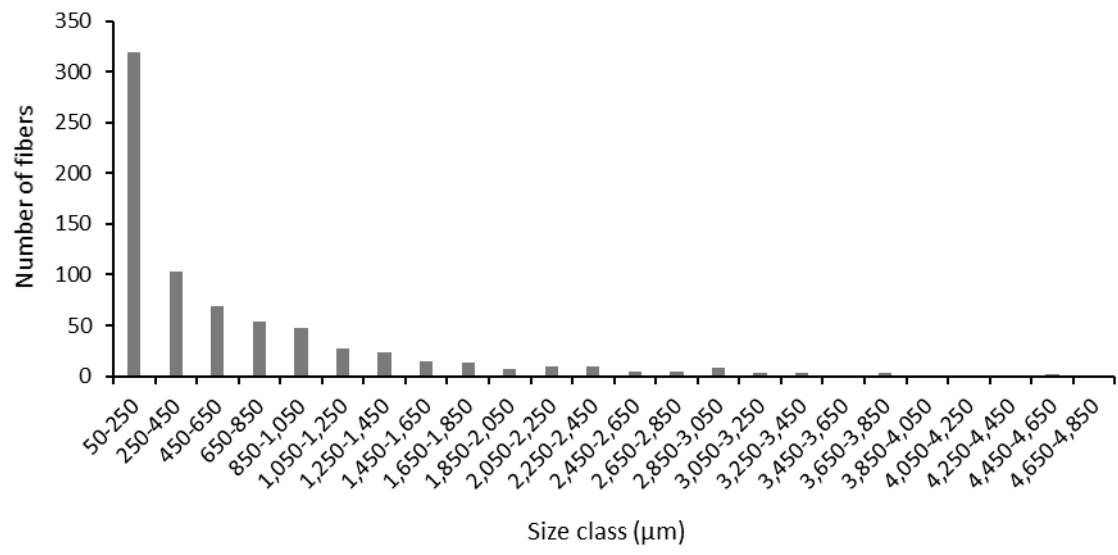


Figure 4: Cumulative anthropogenic fiber size distribution for all sites (n=7).

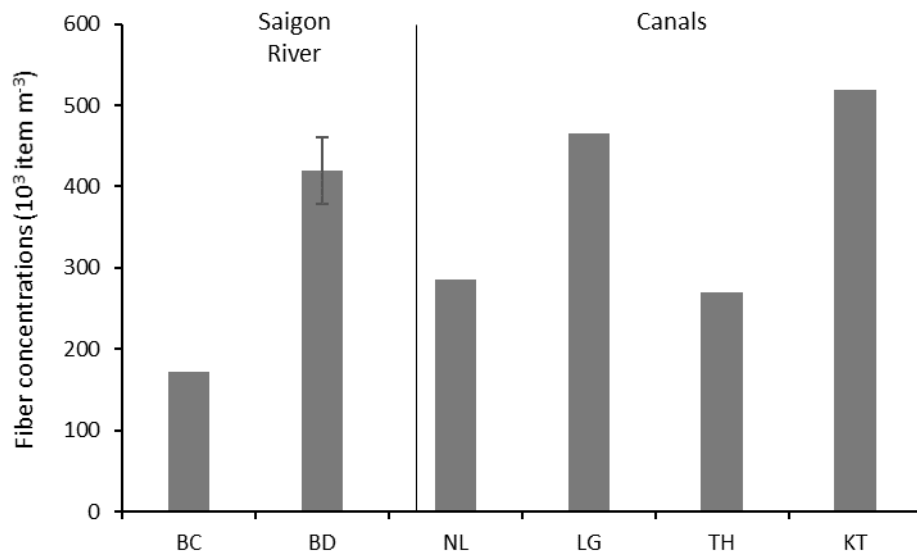


Figure 5: Anthropogenic fiber concentrations calculated in the Saigon River and canals (n=2 for BD).

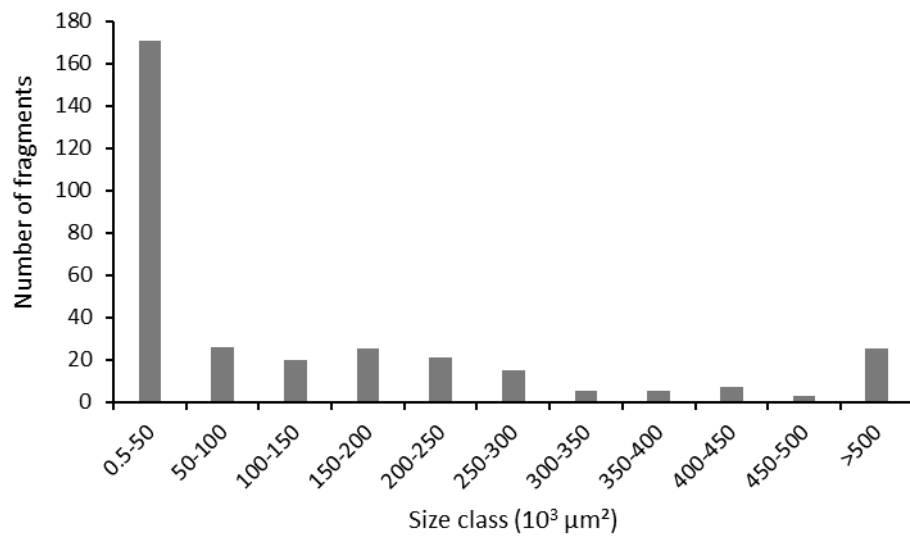


Figure 6: Cumulative fragment size distribution for all sampling sites (n=6).

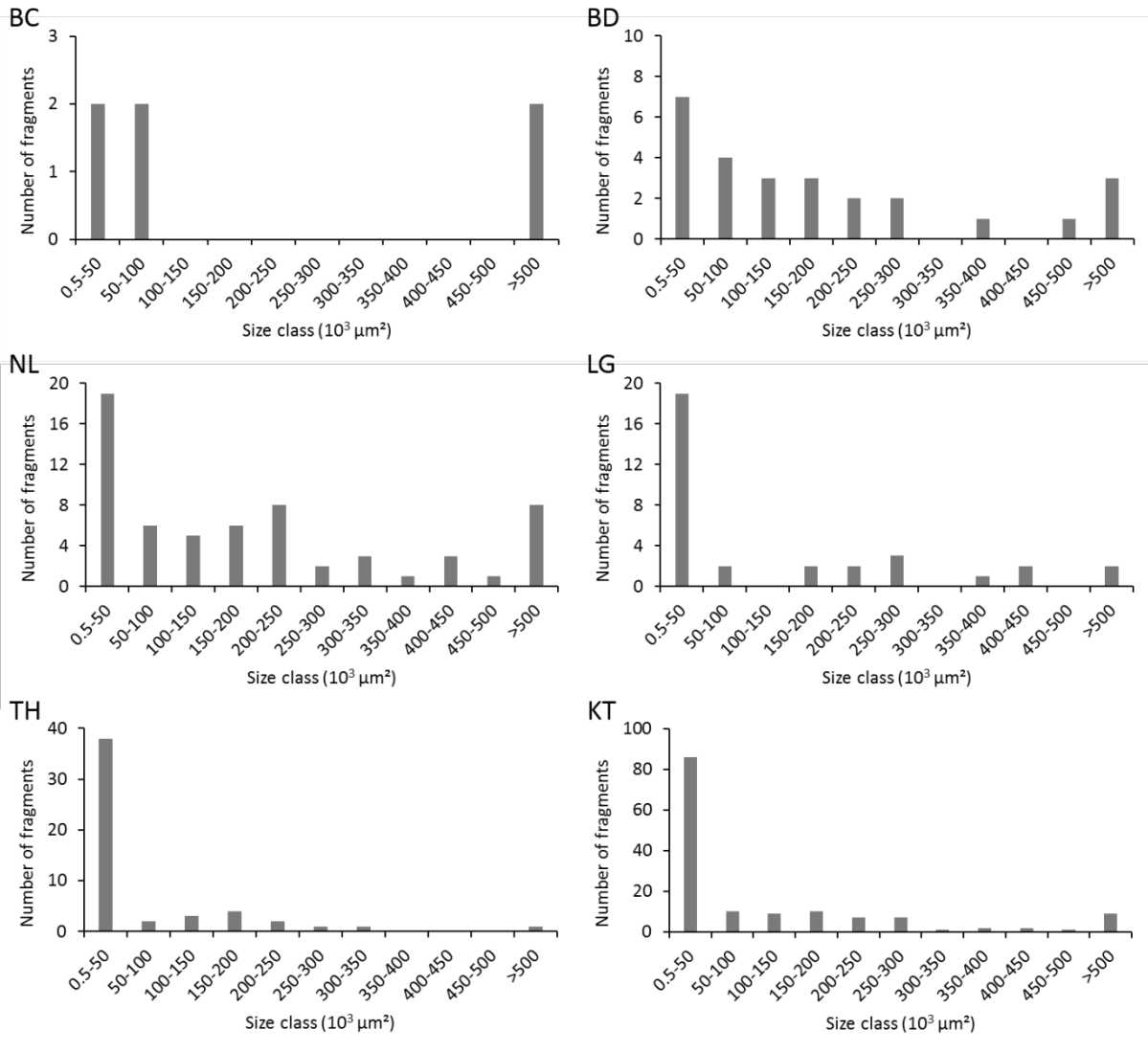


Figure 7: Fragment surface distribution at each site (n=6).

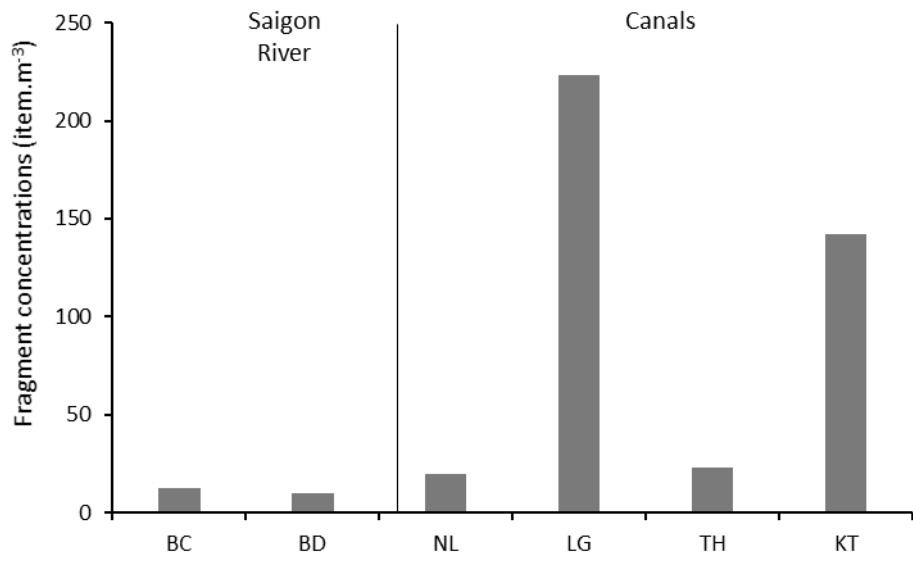


Figure 8: Fragment concentrations in the Saigon River and canals (n=1 for each site)

Size class	Sites							Cumulative
	BC	BD (1)	BD (2)	NL	LG	TH	KT	
Fibers (μm)	BC	BD (1)	BD (2)	NL	LG	TH	KT	Cumulative
50-250	20	100	93	27	30	13	36	319
250-450	8	7	2	20	23	14	29	103
450-650	5	5	2	8	19	11	19	69
650-850	2	1	6	10	13	5	17	54
850-1,050	6	6	5	7	7	8	9	48
1,050-1,250	2	4	5	1	4	4	7	27
1,250-1,450	3	1	0	3	10	3	3	23
1,450-1,650	3	0	1	2	4	2	2	14
1,650-1,850	0	0	1	1	5	3	3	13
1,850-2,050	1	0	0	0	2	3	1	7
2,050-2,250	0	2	1	0	5	0	2	10
2,250-2,450	0	1	0	0	3	4	2	10
2,450-2,650	0	1	0	0	2	0	1	4
2,650-2,850	0	1	0	0	1	0	2	4
2,850-3,050	1	0	0	0	4	0	3	8
3,050-3,250	0	0	0	1	1	0	1	3
3,250-3,450	0	0	0	0	1	0	2	3
3,450-3,650	0	0	0	0	1	0	0	1
3,650-3,850	0	0	0	0	0	2	1	3
3,850-4,050	0	0	0	0	0	0	0	0
4,050-4,250	0	0	0	0	0	0	0	0
4,250-4,450	0	0	0	0	0	0	0	0
4,450-4,650	0	0	0	1	0	1	0	2
4,650-4,850	0	0	0	0	0	0	0	0
Fragments ($10^3 \mu\text{m}^2$)	BC	BD		NL	LG	TH	KT	Cumulative
0.5-50	2	7		19	19	38	86	171
50-100	2	4		6	2	2	10	26
100-150	0	3		5	0	3	9	20
150-200	0	3		6	2	4	10	25
200-250	0	2		8	2	2	7	21
250-300	0	2		2	3	1	7	15
300-350	0	0		3	0	1	1	5
350-400	0	1		1	1	0	2	5
400-450	0	0		3	2	0	2	7
450-500	0	1		1	0	0	1	3
>500	2	3		8	2	1	9	25

Table 1: **Anthropogenic fiber** and fragment size distribution for the six sites (two sampling at BD).

Supporting information

Macro and microplastic contamination assessment of a tropical river (Saigon River, Vietnam) transversed by a developing megacity

Lisa Lahens^{a,c}, Emilie Strady^{b,c*}, Thuy-Chung Kieu-Le^c, Rachid Dris^a, Kada Boukerma^d,
Emmanuel Rinnert^d, Johnny Gasperi^{a*}, Bruno Tassin^{a*}

- a. Université Paris-Est, LEESU, UMR MA 102 - AgroParisTech, 61 avenue du Général de Gaulle, 94010 Créteil Cedex, France
- b. Univ. Grenoble Alpes, CNRS, IRD, Grenoble INP, IGE, F-38000 Grenoble, France
- c. CARE, Ho Chi Minh City University of Technology, VNU-HCM, Viet Nam
- d. Ifremer, LDCM, Centre Bretagne, ZI de la Pointe du Diable, CS 10070, 29280 Plouzané, France

* Corresponding authors: e-mail: emilie.strady@ird.fr, gasperi@u-pec.fr, bruno.tassin@enpc.fr



Figure S1: Pictures of debris collection on Nhieu Loc – Thi Nghe canal by the municipal company.

Table S1: Estimations of the [daily land-based macroplastic waste entering the river](#) per inhabitant ($\text{g inhabitant}^{-1} \text{day}^{-1}$) on Nhieu Loc – Thi Nghe canal based on daily captured debris weight data. The estimations are based on four [assumptions](#) (refer to 2.2 Macroplastic assessment) relative to the daily captured debris weight, the efficiency of capture, the percentage of plastic items and the basin population.

Captured debris weight (tons day^{-1})	Capture %	Canal debris weight (tons day^{-1})	Plastic %	Plastic weight (tons day^{-1})	Basin population (inhabitants)	land-based macroplastic waste entering the river ($\text{g inhabitant}^{-1} \text{day}^{-1}$)
10	100	10	11.5	1.15	1200000	0.96
			522000	2.20		
			23.3	2.33	1200000	1.94
			522000	4.46		
			43.3	4.33	1200000	3.61
			522000	8.30		
	50	20	11.5	2.3	1200000	1.92
			522000	4.41		
			23.3	4.66	1200000	3.88
			522000	8.93		
			43.3	8.66	1200000	7.22
			522000	16.59		
12	100	12	11.5	1.38	1200000	1.15
			522000	2.64		
			23.3	2.796	1200000	2.33
			522000	5.36		
			43.3	5.196	1200000	4.33
			522000	9.95		
	50	24	11.5	2.76	1200000	2.30
			522000	5.29		
			23.3	5.592	1200000	4.66
			522000	10.71		
			43.3	10.392	1200000	8.66
			522000	19.91		

Table S2: Estimations of fiber mass concentrations based on four [assumptions](#) relative to the minimum and maximum diameters of fibers (ϕ : 8 to 20 μm) and the minimum and maximum densities of polymer (ρ : 0.9 to 1.4 g cm^{-3}).

Site	Sample volume (m^3)	Cumulative length (μm)	Fibers mass concentrations (mg.m^{-3})				
			min ϕ - min ρ	min ϕ - max ρ	max ϕ - min ρ	max ϕ - max ρ	Median
			8 0.9	8 1.4	20 0.9	20 1.4	
BC	2.96×10^{-4}	3.12×10^4	4.76	7.41	29.78	46.32	18.60
BD (1)	2.90×10^{-4}	4.00×10^4	6.23	9.70	38.96	60.61	24.33
BD (2)	3.00×10^{-4}	3.13×10^4	4.72	7.35	29.52	45.92	18.43
NL	2.84×10^{-4}	4.54×10^4	7.24	11.26	45.25	70.394	28.26
LG	2.90×10^{-4}	1.39×10^5	21.62	33.63	135.13	210.20	84.38
TH	2.70×10^{-4}	6.74×10^4	11.30	17.57	70.60	109.82	44.08
KT	2.70×10^{-4}	1.35×10^5	22.69	35.30	142.83	220.63	88.57

Table S3: Estimations of fragment mass concentrations based on four [assumptions](#) relative to the minimum and maximum surfaces of fragments (μ : 10 to 40 μm^2) and the minimum and maximum densities of polymer (ρ : 0.9 to 1.6 g cm^{-3}).

Site	Sample volume (m^3)	Cumulative surface (μm^2)	Fragments mass concentrations (mg.m^{-3})				Median
			min μ - min ρ	min μ - max ρ	max μ - min ρ	max μ - max ρ	
			10 0.9	10 1.6	40 0.9	40 1.6	
BC	0.47	2.0×10^6	0.04	0.06	0.15	0.24	0.11
BD	2.69	6.6×10^6	0.02	0.03	0.09	0.14	0.06
NL	3.18	1.8×10^7	0.05	0.08	0.20	0.32	0.14
LG	0.15	5.0×10^6	0.31	0.48	1.23	1.91	0.85
TH	2.28	3.1×10^6	0.01	0.02	0.05	0.08	0.03
KT	1.01	1.8×10^7	0.16	0.26	0.66	1.02	0.46