



HAL
open science

Abundance, composition and fluxes of plastic debris and other macrolitter in urban runoff in a suburban catchment of Greater Paris

Robin Treilles, Johnny Gasperi, Saad Mohamed, Romain Tramoy, Jérôme Breton, Alain Rabier, Bruno Tassin

► To cite this version:

Robin Treilles, Johnny Gasperi, Saad Mohamed, Romain Tramoy, Jérôme Breton, et al.. Abundance, composition and fluxes of plastic debris and other macrolitter in urban runoff in a suburban catchment of Greater Paris. *Water Research*, 2021, 192, pp.116847. 10.1016/j.watres.2021.116847. hal-03113127

HAL Id: hal-03113127

<https://hal-enpc.archives-ouvertes.fr/hal-03113127>

Submitted on 13 Feb 2023

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

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



Distributed under a Creative Commons Attribution - NonCommercial | 4.0 International License

1 **Abundance, composition and fluxes of plastic debris and other macrolitter in urban**
2 **runoff in a suburban catchment of Greater Paris**

3

4 TREILLES Robin^{1*}, GASPERI Johnny^{1,2}, SAAD Mohamed¹, TRAMOY Romain¹, BRETON
5 Jérôme³, RABIER Alain³, TASSIN Bruno¹

6

7 ¹ LEESU, Ecole des Ponts, Univ Paris Est Créteil, Marne-la-Vallée ou Créteil, France

8

9 ² GERS-LEE Université Gustave Eiffel, IFSTTAR, F-44344 Bouguenais, France

10

11 ³ Direction des Services de l'Environnement et de l'Assainissement du Val-de-Marne
12 (DSEA), Conseil départemental du Val-de-Marne, Créteil, France

13

14 *Corresponding author: robin.treilles@enpc.fr

15

16 Abstract

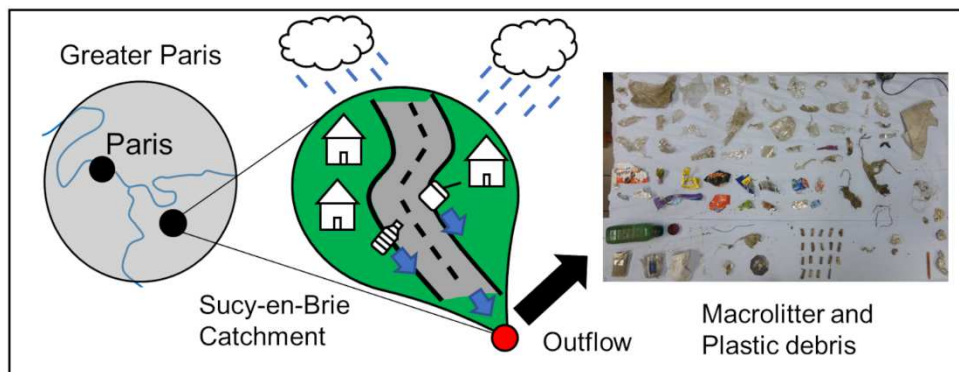
17 Stormwater possibly represents a significant input for plastic debris in the environment;
18 however, the quantification and composition of plastic debris and other macrolitter in
19 stormwater are not available in literature and the amounts discharged into freshwater have
20 been poorly investigated. To obtain a better understanding, the occurrence, abundance, and
21 composition of the macrolitter in screened materials from stormwater were investigated at a
22 small residential suburban catchment (Sucy-en-Brie, France) in Greater Paris. The
23 macrolitter, particularly the plastic debris, was sorted, weighed, and classified based on the
24 OSPAR methodology. On average, plastics accounted for at least 62% in number and for

25 53% of the mass of all the anthropogenic waste found in the screened materials. The most
26 common items were plastic bags or films, crisp or sweet packets, cigarette butts, plastic
27 fragments of unknown origin, garbage bags or garbage bag strings, foil wrappers, tampon
28 applicators, plastic cups, and medical items such as bandages. Plastic debris concentrations
29 in runoff water ranged between 7 and 134 mg/m³ (i.e. 0.4–1.7 kg.yr⁻¹.ha⁻¹ or 4.8–18.8 g.yr⁻¹
30 .cap⁻¹). When extrapolated to the Greater Paris area, the estimated amount of plastic debris
31 discarded into the environment through untreated stormwater of separate sewer systems
32 ranges from 8 to 33 tons yr⁻¹.

33

34 KEYWORDS: macrolitter, plastic debris, stormwater, urban inputs

35 Graphical abstract



36

37 1. Introduction

38 For several years, studies have demonstrated the strong environmental impacts of plastic
39 debris on marine (Barnes, 2002; Derraik, 2002; Gall and Thompson, 2015) and freshwater
40 (Blettler et al., 2017) ecosystems. However, recent field studies (van Emmerik et al., 2018)
41 and models (Lebreton et al., 2017; Schmidt et al., 2017) have shown that rivers originating
42 from populated metropolitan areas represent a major source of the plastic pollution in
43 oceans. Additionally, the existence and performance of solid waste management practises
44 and sewer systems play a key role in plastic waste discharge (Blettler et al., 2018; Jambeck
45 et al., 2015).

46 Most plastic pollution studies focus on microplastics (<5 mm) which correspond to the most
47 numerous debris discarded in the environment. However, macroplastics (>5 mm) account for
48 the most significant fraction in terms of mass (Van Sebille et al., 2015). In this study, plastic
49 debris only includes macroplastics. The understanding of macrolitter and plastic debris is still
50 inadequate (Blettler et al., 2018) and discrepancies between plastic emission models and
51 field data have been reported in several studies (Blettler et al., 2018; González-Fernández
52 and Hanke, 2017; Schöneich-Argent et al., 2020; Tramoy et al., 2019b); therefore, additional
53 field data in urban areas should be collected to reduce these discrepancies. The role and
54 importance of urban areas in the generation and transfer of plastic debris have been
55 identified and frequently mentioned in previous studies; however, studies and data that
56 precisely assess the role of these complex sources on plastic pollution are minimal.

57 Plastic debris, primarily microplastics, has been reported in every type of urban water source
58 including the atmosphere and rainwater (Chen et al., 2020; Dris et al., 2016), drinking water
59 (Mintenig et al., 2019; Pivokonsky et al., 2018), wastewater entering treatment plants
60 (WWTPs) and in effluents (Magni et al., 2019; Talvitie et al., 2015), sludge (Li et al., 2018;
61 Mintenig et al., 2017), and stormwater (Dris et al., 2018; Liu et al., 2019; Piñon-Colin and al.,
62 2020). However, the effects of the dynamics, abundance, and composition of macrolitter on
63 an urban scale and its consequences on the receiving hydrosystem are poorly understood.

64 No comprehensive approach can precisely describe the plastic debris in urban environments
65 or facilitate the design of a conceptual quantitative model of plastic fluxes in urban areas.

66 The high variability of the results and the lack of clear explanatory factors impede the ability
67 to derive definitive conclusions on macrolitter, particularly plastic debris fluxes (Blettler et al.,
68 2018). This study focused on the plastic debris fluxes in the urban runoff at the outlet of a
69 small urban catchment in a Paris suburb.

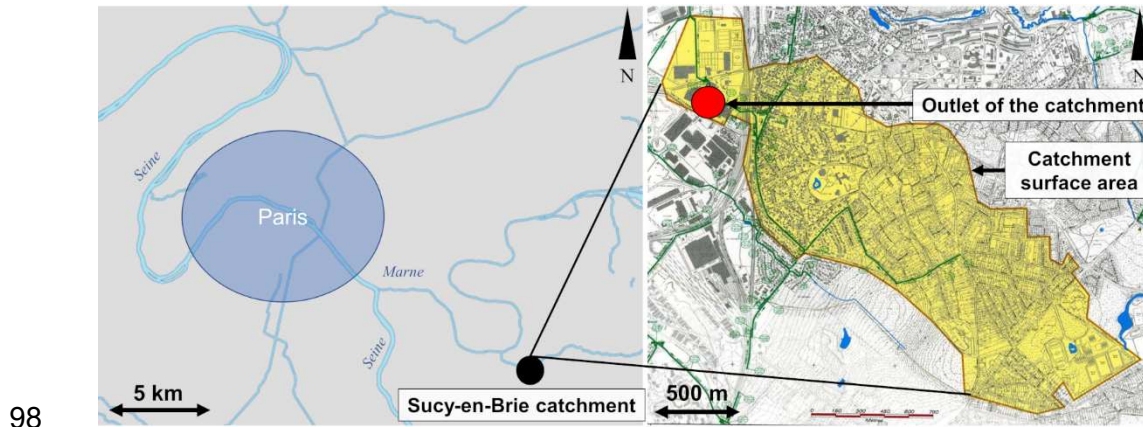
70 This study aims to (i) provide data on the composition of the macrolitter in the runoff water of
71 a small urban catchment; (ii) assess the mass percentages of macrolitter, particularly plastic

72 debris; and (iii) estimate the plastic debris mass fluxes per hectare of impervious area and
73 per capita and extrapolate those figures to the scale of Greater Paris.

74 2. Materials and methods

75 2.1. Sampling site

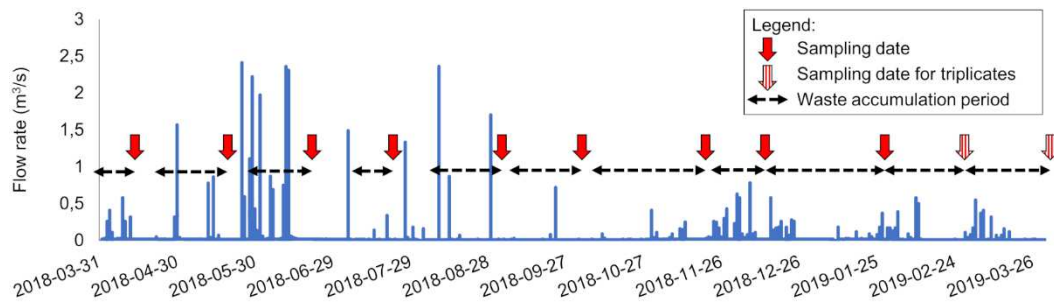
76 Samples were collected at the outflow of the Sucy-en-Brie watershed, which were located in
77 a suburban environment in the southeast portion of the Paris agglomeration (Figure 1). It has
78 a surface area of 228 ha with an impervious area of 62 ha, which represents 27% of the
79 catchment (Gasperi et al., 2017). The population of the territory is approximately 5,700,
80 which is mostly residential, with an individual household density of approximately 25 cap.ha⁻¹
81 that corresponds to a moderately dense urban area in France (Gasperi et al., 2017).
82 Commercial and professional activities are limited. The sewer system in this catchment is a
83 separated one, *i.e.* wastewater and stormwater are collected separately. A stormwater
84 treatment structure is located at the catchment outflow, which consists of a stormwater
85 retention pond and a lamellar settling tank. To block larger debris from entering the
86 treatment structure, a 6 cm screen (S_{6cm}) and a 1 cm screen (S_{1cm}) are installed in upstream
87 retention ponds. This type of stormwater treatment structure of separate sewer system is
88 rare in Greater Paris and crucial for our experiments as it traps macrolitter from Sucy-en-Brie
89 catchment. Debris collected by these screens is automatically deposited into trash
90 containers (one container per screen), which enables the screened materials to be
91 differentiated by the type of screen. The accumulated debris on the two screens was used in
92 this study to investigate macrolitter abundance and composition. Additionally, the stormwater
93 treatment structure is well-instrumented for urban water study. Stormwater flow rates and
94 volumes through the screens were measured by utilizing flowmeters (DRUCK-PTX1830 and
95 DRUCK-PTX5032) and provided by the Val-de-Marne Environmental and Sanitation
96 Services Directorate (DSEA); these measurements were utilized to estimate the macrolitter
97 concentrations.



99 Figure 1: Location of the Sucy-en-Brie catchment. The outlet and stormwater retention pond
100 are located in the western portion of the catchment.

101 2.2. Sampling method

102 Eleven sampling campaigns were performed between April 2018 and April 2019 to collect
103 the screened materials from S_{6cm} and S_{1cm} under different hydrological conditions (Figure 2).
104 During each campaign, samples of the screened materials accumulated in trash containers
105 of each of the screens were collected and weighed, and the initial waste volume for each
106 trash container was estimated before and after sampling. The densities of the samples were
107 then estimated using volume and weight. The samples were homogenised, and a subsample
108 was randomly collected and weighed (~10% of the initial sample mass, which corresponds
109 to 3–6 kg). The subsamples were then dried and sorted to study the variations in the
110 macrolitter and plastic compositions (see Section 3). The last two campaigns were
111 performed in triplicate to study intra-sample variability and to assess the robustness of the
112 analytical procedure.



113

114 Figure 2: Stormwater hydrogram of the Sucy-en-Brie catchment and sampling dates. Waste
 115 accumulation period for each sample is indicated.

116 2.3. Analytical procedure

117 The collected debris had a high water content (>70% of the initial mass); therefore, the
 118 subsamples were dried in an oven at 40°C for at least 10 d, after which the dry debris was
 119 weighed and visually sorted. The first four campaigns focused only on plastic waste and
 120 cigarette butts; however, all during the following campaigns other anthropogenic items
 121 (aluminium cans, healthcare waste, etc.) larger than 5 mm were classified using the OSPAR
 122 classification (OSPAR Commission, 2010). Additionally, items were weighed according to their
 123 waste category: plastics, metals, sanitary and medical waste, and other anthropogenic waste
 124 (composite waste, glass, cardboard, etc.). In this study, sanitary and medical waste included
 125 items in OSPAR classifications 97, 98, 99, 100, 102, and 105. For the plastics category, only
 126 synthetic materials were considered. Artificial and composite materials were considered
 127 separately to enable a better distinction between materials; therefore, cigarette butts were
 128 not included in the plastic category. An additional category; “non-plastic anthropogenic
 129 waste” has been defined as all anthropogenic waste excepted plastic items which combines
 130 metals, sanitary and medical waste, and other anthropogenic waste.

131 Using the stormwater volumes, the mass percentages of the different subsamples were
 132 extrapolated to the initial debris volume to estimate plastic debris concentrations in the
 133 stormwater.

134 2.4. Calculation of plastic debris flux in stormwater

135 Two methods were used to estimate the annual plastic debris mass in the screened
136 materials, namely, (i) using the estimated plastic debris concentration in stormwater and the
137 annual stormwater volume (method_{Concentration}) and (ii) using the mean tonnage of the
138 screened materials accumulated from 2015 to 2019 and the mean plastic mass percentage
139 estimated by this study (method_{Annual Mass}).

140 For method_{Concentration}, the results of the analytical procedure presented in Section 3 were
141 used to calculate the plastic debris concentrations in the stormwater (N = 11). The mean and
142 median values were then multiplied by the annual stormwater volume filtered through the
143 screens (from April 2018 to April 2019); consequently, the plastic debris mass in the
144 screened materials was obtained.

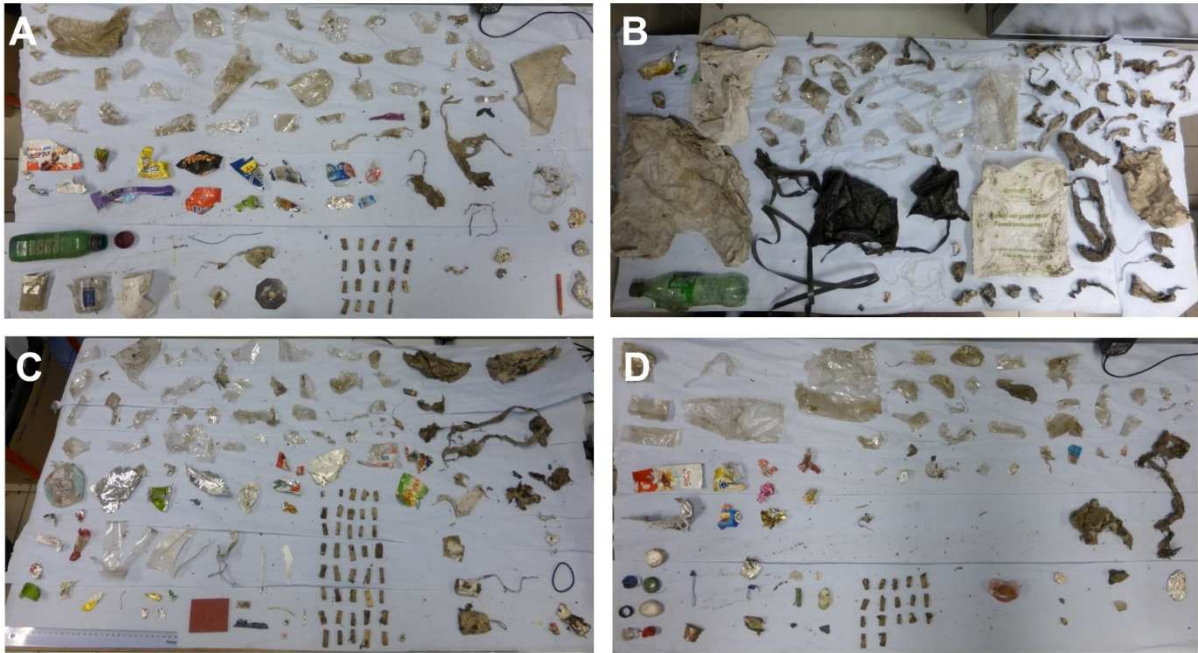
145 For method_{Annual Mass}, waste mass percentages in the subsamples were directly applied to the
146 annual tonnage of the screened materials collected by a company responsible for its
147 incineration. For this study, it was assumed that the plastic mass percentage was constant
148 over the last five years and the DSEA provided screened materials tonnage estimations from
149 2015 to 2019.

150 The plastic debris masses determined by both methods were then normalised to the
151 impervious surface area of the catchment and population, which yielded two different ratios,
152 ratio_{Area} and ratio_{Cap} expressed in $\text{kg}\cdot\text{yr}^{-1}\cdot\text{ha}^{-1}$ and $\text{g}\cdot\text{yr}^{-1}\cdot\text{cap}^{-1}$, respectively.

153 3. Results

154 3.1. Macrolitter composition in screened materials

155 Figure 3 illustrates the different waste types and categories that were collected during the
156 campaigns. The anthropogenic macrolitter composition of the screened materials is
157 presented in Figure 4. All items found at each screen are presented in the supplementary
158 data (Table S1 and S2). In this paragraph, percentages will only refer to percentages in
159 numbers and not in mass.

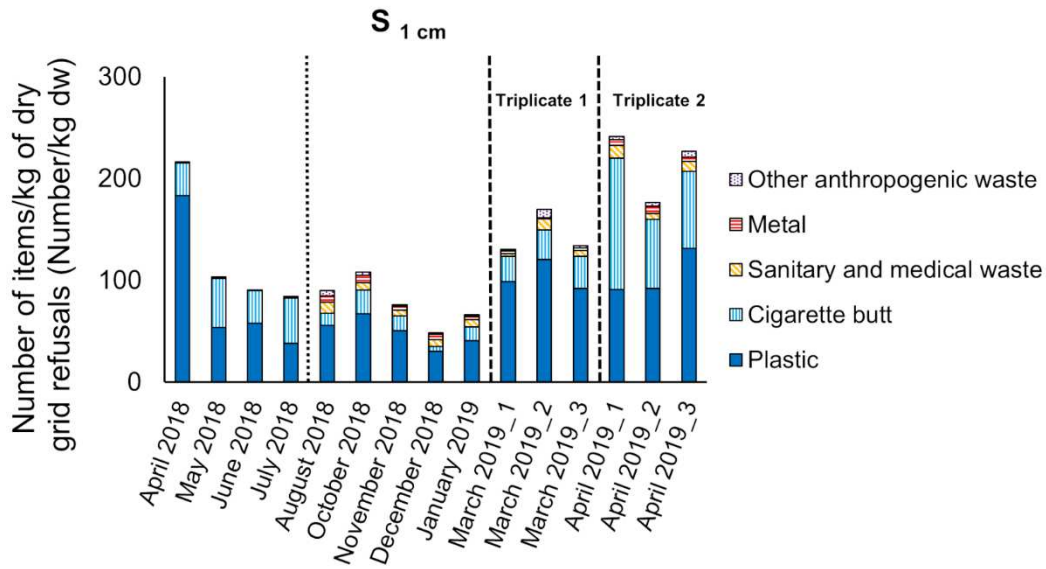
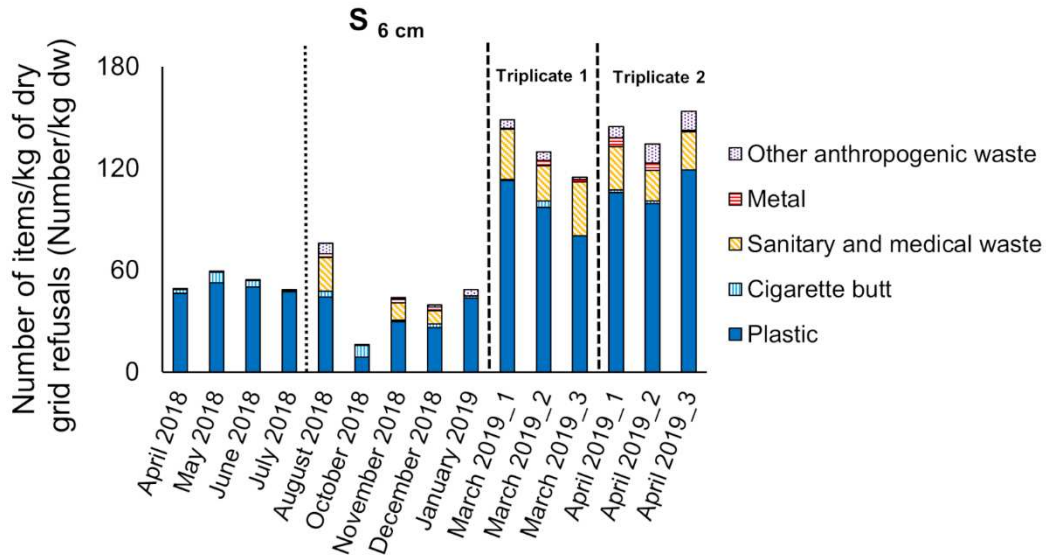


160

161 Figure 3: Common waste found in S_{6cm} (A and B) and S_{1cm} (C and D)

162 Only anthropogenic waste was included in Figure 3 and Figure 4. Natural organic debris
 163 (plant debris and putrescible waste) was not categorised in detail and only weighed (*c.f.* §2).

164 For S_{6cm} and S_{1cm} , the plastic category was the most numerous with mean values of $71\pm 9\%$
 165 and $62\pm 10\%$ ($N = 11$ with triplicates), respectively, excluding the first four campaigns. For
 166 S_{6cm} , medical and sanitary waste had the second-largest percentage ($16\pm 9\%$) and consisted
 167 mainly of bandages. For S_{1cm} , cigarette butts had the second-largest percentage ($24\pm 13\%$).
 168 Other material types (paper/cardboard, metal, etc.) accounted for the smallest percentage
 169 ($< 7\%$). For S_{6cm} and S_{1cm} , both triplicates showed a relatively low variability for the plastic
 170 category (variation between the minimum and maximum values was $< 8\%$ and $< 34\%$ for S_{6cm}
 171 and S_{1cm} , respectively).



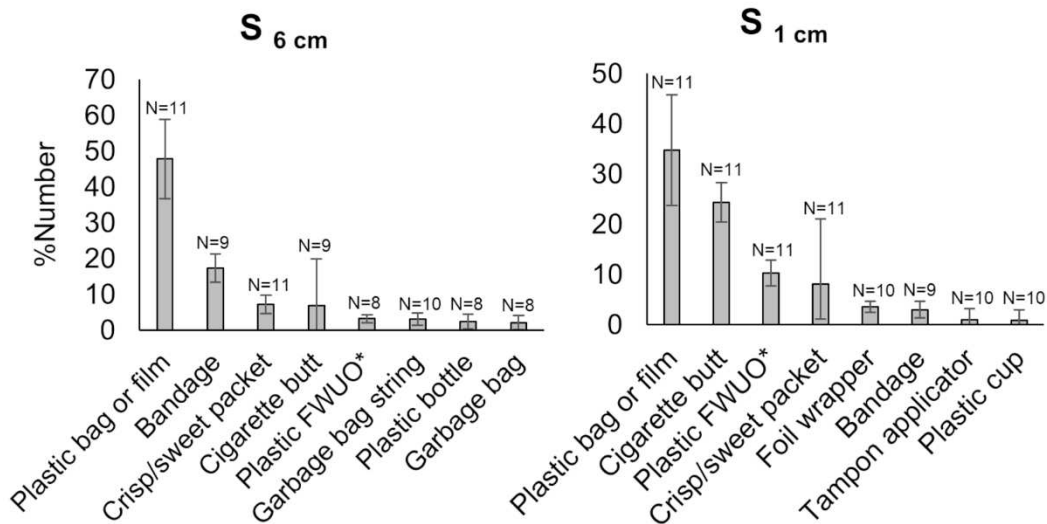
172

173 Figure 4: Anthropogenic macrolitter composition for each screen. The first four campaigns
 174 (April - July 2018 are separated by a dotted line) only focused on plastics and cigarette butts.

175 The y-axis is different for each graph. Triplicates 1 and 2 are separated by dashed lines.

176 To characterise the plastic pollution in the stormwater, the most common items found in S_{6cm}

177 and S_{1cm} (Figure 5) were identified.



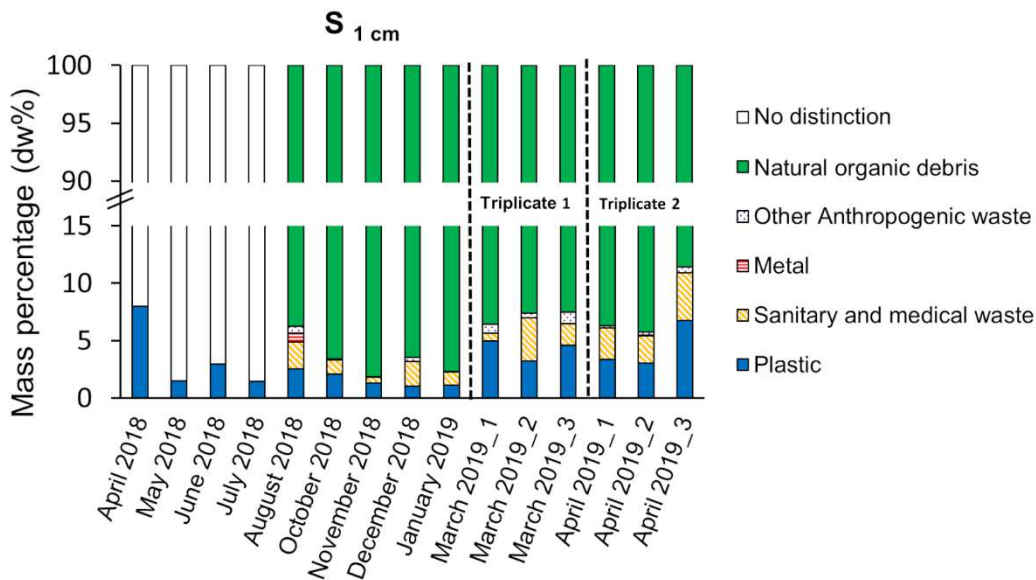
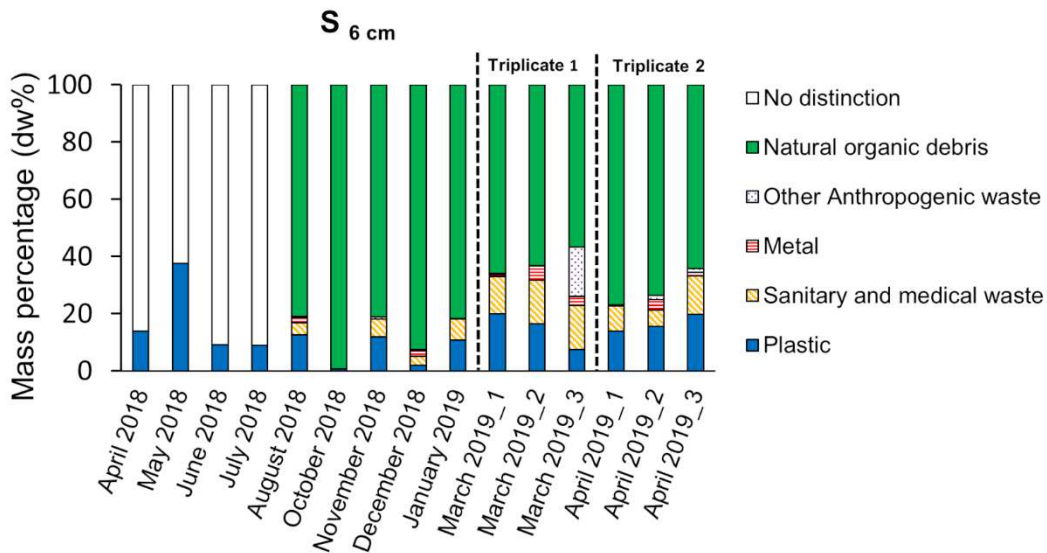
178

179 Figure 5: Mean percentages of the eight most common items found in the screened
 180 materials. The error bars illustrate the standard deviations and N denotes the number of
 181 samples where the item was present. For a more accurate comparison, the first four
 182 campaigns were not included. *Plastic FWUO = plastic fragment with unknown origin

183 Plastic bags and films, cigarette butts and bandages were the most numerous items found in
 184 the screened materials samples (Figure 5). Plastic bags and films were the predominant
 185 items found in S_{6cm} and S_{1cm} of all the other items. The most common items found in S_{6cm}
 186 and S_{1cm} are similar; however, they do not account for the same proportions.

187 3.2. Macrolitter mass percentages in screened materials and concentrations in urban
 188 runoff

189 Percentages by dry weight (dw%) of each waste category for each screen are presented in
 190 Figure 6. The highest average percentages for S_{6cm} and S_{1cm} corresponded to natural
 191 organic debris (76±13 and 94±3 dw%, respectively), plastics (12±6 and 3±2 dw%,
 192 respectively), and sanitary and medical waste (8±5 and 2±1 dw%, respectively) with N = 11
 193 (with triplicates) and the first four campaigns were not included in the mean values. Other
 194 anthropogenic waste (2±5 and <1 dw% for S_{6cm} and S_{1cm}, respectively) and metals (2±2 and
 195 <1 dw%, respectively) accounted for minor percentages, except for one sample (March
 196 2019_3).



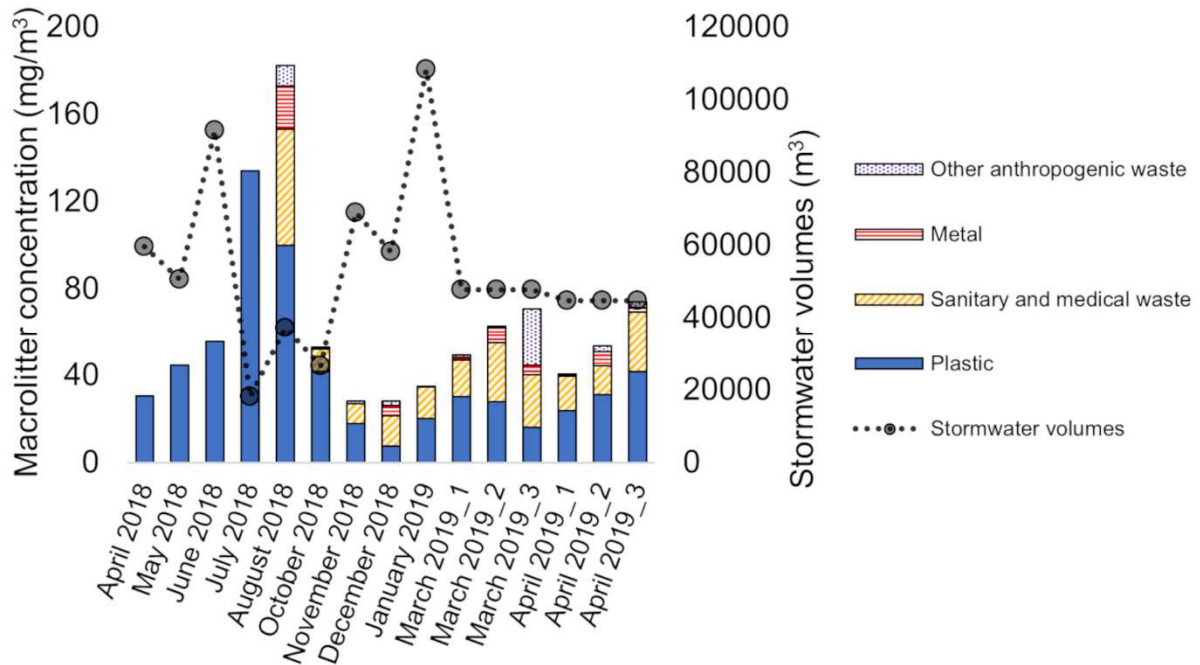
197

198 Figure 6: Percentages by dry weight (dw%) of each waste category for each screen. Only
 199 plastics and cigarette butts were included in the first four samples.

200 For triplicates 1 and 2 at S_{6cm} , plastic mass percentage ranges were 7-20 and 14-20 dw%,
 201 respectively, whereas for triplicates 1 and 2 at S_{1cm} , the ranges were 3-5 and 3-7 dw%,
 202 respectively. When all the anthropogenic waste was compared for triplicates 1 and 2 at S_{6cm}
 203 (plastics, metals, sanitary and medical waste, and other anthropogenic waste) the mass
 204 percentage ranges were 34-43 and 23-35 dw%, respectively, and for triplicates 1 and 2
 205 S_{1cm} , these ranges were 6-7 and 6-11 dw%.

206 3.3. Plastic debris flux

207 The macrolitter concentration of stormwater (mg/m³, Figure 7) was calculated based on the
 208 collected data.



209

210 Figure 7: Macrolitter concentrations (mg/m³) and stormwater volumes filtered through the
 211 screens for the studied periods (both screens S_{6cm} and S_{1cm} are cumulated)

212 The concentrations of all the anthropogenic waste ranged from 28 to 182 mg/m³ and the
 213 mean and median concentrations of each waste category are presented in Table 1. Mean
 214 values are always higher than median values owing to heavy items that impact the mean
 215 values.

216 Table 1: Mean and median concentrations for each waste category (N = 15 for plastics and
 217 11 for other categories)

	Mean concentration ± standard deviation (mg/m ³)	Median concentration (mg/m ³)
Plastic	41±33	31
Sanitary and medical waste	21±13	16
Metal	4±6	2
Other anthropogenic waste	4±8	1
Natural organic debris	811±1445	247

218 The natural organic debris concentrations are not presented in Figure 7 because their
 219 concentrations are significantly higher than the other waste categories. The plastic debris
 220 concentrations ranged between 7 and 134 mg/m³ (minimum and maximum values,
 221 respectively).

222 Utilizing the method_{Concentration}, the mean and median mass of the plastic debris accumulated
 223 on the screens in one year were 27±22 and 21 kg, respectively.

224 For the method_{Annual Mass}, major fractions found in the screened materials and the percentage
 225 by weight (w%) of plastics accumulated on both screens (estimated from mass percentages
 226 previously presented) are summarised in Table 2.

227 Table 2: Mean composition of screened materials and estimation of mean plastic mass
 228 accumulated in one year on the screens (mean value ± standard deviation)

	S _{6cm} and S _{1cm} combined
Water content (w%)	74±4
Organic waste mass (w%)	22±4
Plastic and non-plastic anthropogenic waste mass (w%)	4±2
Plastic waste mass (w%)	2±1
Total mass of screened materials per year (mean value from 2015 to 2019, kg)	5,359±667
Estimation of plastic mass per year in screened materials (mean value from 2015 to 2019, kg)	107±55

229

230 Based on this data, 107±55 kg of plastic debris were accumulated in the screened materials
 231 of Sucy-en-Brie in one year.

232 The results of these two methods can be normalised to the impervious surface area (62 ha)
 233 and population (~5,700 inhabitants) of Sucy-en-Brie to calculate ratio_{Area} and ratio_{Cap},
 234 respectively, which are provided in Table 3.

235 Table 3: Annual plastic debris flux normalized to impervious surface area and population of
 236 Sucy-en-Brie for method_{Concentration} and method_{Annual Mass}

Sucy-en-Brie	Method _{Concentration}	Method _{Annual Mass}
Annual plastic flux in stormwater of Sucy-en-Brie (kg.yr ⁻¹)	27.4±22	107.2±55.2
Ratio _{Area} : plastic flux per impervious surface area (kg.yr ⁻¹ .ha ⁻¹)	0.4±0.3	1.7±0.9
Ratio _{Cap} : plastic flux per capita (g.yr ⁻¹ .cap ⁻¹)	4.8±3.9	18.8±9.7

237

238 4. Discussion

239 4.1. Macrolitter composition in screened materials

240 Because they are in series, differences in the waste composition of the S_{6cm} and S_{1cm}
241 screened materials can be observed (Figure 3 and 5), which is attributed to the mesh size
242 difference. The most important difference in waste composition is the abundance of cigarette
243 butts in the S_{1cm} material. Generally, cigarette butts pass through S_{6cm} but not through S_{1cm}.
244 The S_{1cm} mesh size is not small enough to retain all the cigarette butts in the stormwater, as
245 evidenced by the presence of cigarette butts in the lamellar settling tank (personal
246 observation); however, the fraction that is not retained is difficult to estimate. Based on their
247 distinctive shape, some plastic films were determined to be discarded cigarette box
248 packaging.

249 This study found 52 and 60 different item categories and 1,613 and 3,126 items for S_{6cm}
250 (Table S1) and S_{1cm} (Table S2), respectively. Plastic debris represented 71% and 62% of the
251 S_{6cm} and S_{1cm} items, respectively, which reflects the relatively low diversity of the
252 composition of the screened materials and the predominance of plastic waste. Plastic bags
253 and films were the most common items found in the screened materials. Bandages were
254 also common, which could be related to the proximity of health facilities to the catchment,
255 mismanagement of health and sanitary waste, and illicit disposal; this is because this type of
256 waste requires costly disposal procedures. Because condoms and sanitary napkins were
257 observed in the waste, misconnections between the stormwater and wastewater systems

258 most likely exist in this catchment. These misconnections are easily identified in separate
259 sewer systems (Ellis and Butler, 2015). The most recent estimate is that 10% of all
260 connections are misconnections between stormwater and wastewater sewers (data provided
261 by the DSEA), which explains the presence of these types of unexpected waste.

262 Considering the relatively low variability between waste categories and the eight most
263 common items found, the waste composition must be linked to several parameters such as:
264 (i) the habits of the citizens, (ii) the layout of the sewer network (e.g. illicit connections, layout
265 of gully pots) and (iii) the cleaning service of Sucy-en-Brie (e.g. garbage bin availability,
266 urban cleaning). The distribution of the screened materials may reflect the type of items that
267 are socially acceptable to discard in the street, easily lost, or difficult to clean, except for
268 waste caused by errors linked to misconnections (e.g. tampon applicators), illicit disposal to
269 avoid disposal costs (e.g. bandages), and animal behaviour (e.g. birds) that could potentially
270 spread macrolitter. However, additional studies on these topics are necessary to confirm
271 these trends.

272 4.2. Macrolitter and plastic debris mass percentages in screened materials and 273 concentrations in urban runoff

274 When the S_{6cm} and S_{1cm} waste from the same campaigns are combined, the mass of the
275 screened materials is primarily composed of water (>70 w%) and natural organic debris
276 (~22 w%) (Table 2). Non-plastic anthropogenic waste and plastic debris account for 4 ± 2 and
277 2 ± 1 w%, respectively. The plastic debris percentage in the screened materials was low as
278 compared to that of natural organic debris; however, the mass of the plastic debris
279 corresponds to a mean percentage of 53 ± 16 w% of all the anthropogenic waste mass,
280 showing the abundance of plastic debris. Although some waste categories are abundant in
281 number (i.e. cigarette butts), they represent minor mass fractions (Figure 4 and 6).

282 The natural organic debris concentrations showed the highest variability with a standard
283 deviation of $1,145 \text{ mg/m}^3$ and a high variation between the minimum (176 mg/m^3 in March)
284 and maximum values ($4,975 \text{ mg/m}^3$ in October) (Figure 7 and Table 1). This is assumed to

285 be caused by seasonal variability, most likely leaves dropping in autumn that are
286 subsequently transported by the increased precipitation amounts in autumn (Figure 2).
287 Higher anthropogenic waste concentrations, particularly plastic debris concentrations, were
288 observed during the summer period from July to August (Figure 7). Compared to natural
289 organic debris, non-plastic anthropogenic waste and plastic debris presented a different
290 seasonal pattern. Initially, it appears that the plastic debris concentrations correspond to
291 smaller stormwater volumes; however, when plotted against stormwater volume, plastic
292 debris concentration decreases when stormwater volume increases (Figure S3). However,
293 no obvious correlation was found ($R^2 = 0.21$ and $p\text{-value} = 0.08$ utilizing the Spearman-Rs
294 test, Figure S3), which indicates that other parameters influence plastic debris accumulation
295 in the screened materials.

296 Precipitation fluctuations may have a significant influence on plastic debris accumulation. In
297 July and August 2018, only 4 and 5 rain events were recorded, respectively, versus 12–20
298 per month in the winter. The summer and winter periods were compared using the mean
299 stormwater flow rates at the outlet of the catchment for each rain event (Table S4). The July-
300 August rain events presented significantly higher mean flow rates compared to those in the
301 winter period ($p = 0.01$ with a Mann-Whitney-Wilcoxon test, $N = 9$ for the July–August period
302 and $N = 44$ for the winter period). The summer period is characterised by infrequent, intense
303 storm events. High-intensity rain events may carry more waste than less intense rain events;
304 however, the holidays that occur in July and August may cause greater waste discharge due
305 to recreational activities. Both parameters, storm events and holidays, may explain the
306 higher values observed in the July-August period compared to the other periods.

307 4.3. Plastic debris flux

308 As shown in Table 3, the method_{Annual Mass} yields higher mass accumulation values than the
309 method_{Concentration}. Based on the standard deviation of the method_{S Annual Mass} (Table 2), the
310 mass accumulation values are more widespread than those of the method_{Concentration}, which

311 may be because the method_{Annual Mass} uses annual mean values. The application of both
 312 methods enables a better assessment of the plastic accumulation in the screened materials.

313 The Sucy-en-Brie ratios can be extrapolated for the Greater Paris area, which is defined as a
 314 catchment encompassing Paris and 284 neighbouring cities, spanning 183,000 ha, and with
 315 a population of approximately 8.9 million (Risch et al., 2018). Sucy-en-Brie's ratio_{Area} and
 316 ratio_{Cap} were multiplied by the impervious area of Greater Paris (50,900 ha estimated by
 317 Risch et al., 2018) and the Greater Paris population (Table 4). These values correspond to a
 318 maximum plastic litter discharge in the stormwater assuming the habits of the Sucy-en-Brie
 319 citizens, the urban cleaning methods and the layout of the sewer network are representative
 320 of the Greater Paris area. Moreover, these values consider all stormwater, without distinction
 321 of sewer systems (combined or separate). Only a part of this stormwater remains untreated.
 322 To ensure a better comparison between Sucy-en-Brie and Greater Paris, we estimated the
 323 untreated stormwater from separate sewer systems. For this reason, ratio_{Area} was multiplied
 324 by the impervious surface area drained by separate sewer systems (19,000 ha, Table 4).

325 Table 4: Extrapolation of Sucy-en-Brie ratios to the Greater Paris area utilizing
 326 Method_{Concentration} and Method_{Annual Mass}

Greater Paris	Method_{Concentration}	Method_{Annual Mass}
Ratio _{Area} * impervious surface area of Greater Paris (tons.yr ⁻¹)	22.4±17.8	88.1±45.3
Ratio _{Cap} * population of Greater Paris (tons.yr ⁻¹)	42.8±34.6	167.4±86
Ratio _{Area} * impervious surface area connected to separate sewer systems (for untreated stormwater)	8.4±6.6	32.9±12.5

327

328 Using the method_{Concentration} and method_{Annual Mass} and extrapolating the Sucy-en-Brie ratios to
 329 the Greater Paris area, a resultant annual flux of 22–167 metric tons.yr⁻¹ of plastic debris
 330 was calculated. Assuming stormwater of separate sewer systems remains mainly untreated,

331 the plastic debris flux from Greater Paris to the environment through untreated stormwater of
332 separate sewer systems ranges between 8–33 tons.yr⁻¹.

333 The initial study by Tramoy et al. (2019) estimated that the amount of plastic debris
334 discharged from the Seine River to the English Channel ranges between 1,100 and 5,600
335 tons.yr⁻¹, which correspond to 66 and 353 g .cap⁻¹.yr⁻¹, respectively. More recently, Tramoy
336 et al. (2021, in revision) refined their estimations to 6-12 g.yr⁻¹.cap⁻¹, which approximately
337 corresponds to the results of this study, and calculated a plastic debris discharge of
338 approximately 100–200 tons yr⁻¹ into the sea. Other sources may contribute to the plastic
339 debris discharged into the Seine River catchment including combined sewer overflows.

340 Additionally, the plastic discharges attributed to urban traffic may be underestimated. Plastic
341 accumulation along the Seine River has been studied (Tramoy et al., 2019a); however, the
342 precise estimation of plastic debris accumulation is difficult. Gasperi et al. (2014) estimated
343 that ~27 metric tons of plastic are captured annually by floating booms placed downstream
344 of the combined sewer overflows; however, only a portion of the floating debris is captured
345 during storm events.

346 Other factors may influence the plastic debris input into the stormwater, particularly
347 meteorological and hydrological conditions, as determined by van Emmerik et al. (2019) who
348 observed an increase in plastic discharge up to a factor of ten for the Seine River due to
349 meteorological and hydrological conditions. Althoff et al. (2020) estimated the plastic
350 consumption of France to be 70 kg per inhabitant per year. The discarded plastic found in
351 stormwater corresponds to less than 0.3 % (4.8-18.8 g.yr⁻¹.cap⁻¹, Table 3) of the amount
352 consumed per inhabitant. Thus, plastic debris fluxes in stormwater are minimal compared to
353 plastic consumption.

354 However, plastic debris inputs in the Sucy-en-Brie catchment may be higher than what
355 accumulated in the catchment outflow for several reasons. First, municipal street sweeping
356 and sanitation services in Sucy-en-Brie may be effective in preventing most plastic debris
357 from entering in the stormwater. Second, stormwater grates may have prevented the largest

358 size waste from entering the sewers. Third, plastic waste may be retained in sewer systems
359 due to installed structures and obstacles in the sewers. Additionally, the representativity of
360 the Sucy-en-Brie catchment may be discussed, because of its size and limited industrial and
361 commercial activities; therefore, other sites should be studied for comparison. This study,
362 however, provides an initial estimation of the plastic debris in the stormwater of the Greater
363 Paris area. In addition to plastic debris larger than 5 mm, microplastics in stormwater should
364 also be studied to compare the different inputs of macro and microplastics.

365 The results of this study suggest that in urban areas, plastic pollution prevention techniques
366 combining waste collection services and systems (e.g. sanitation services and waste
367 screens to prevent waste from entering the environment) may be effective when performed
368 soon enough. Additionally, plastic waste retention times in the urban areas of developed
369 countries, particularly in sewer systems and on land, might be greater than what is estimated
370 by the models (Lebreton et al., 2017; Schmidt et al., 2017). Additional studies should be
371 performed to compare different urban catchments and confirm these trends.

372 5. Conclusion

373 This study provides the first evaluation of the abundance and composition of macrolitter and
374 plastic debris in stormwater, particularly in screened materials. Screened materials in Sucy-
375 en-Brie are primarily composed of water (~74 w%), natural organic debris (~22 w%), and
376 anthropogenic waste (~4 w%). Among the anthropogenic waste, plastic was the largest in
377 number (>60%) and mass (>50% of anthropogenic waste dry mass, on average). The plastic
378 debris concentration in stormwater ranges from 7 to 134 mg/m³. When extrapolated to the
379 Greater Paris area, discharged plastic debris in stormwater ranged from 22 to 167 tons.yr⁻¹,
380 of which an estimated 8-33 tons yr⁻¹ is discharged into the environment through untreated
381 stormwater from separate sewer systems. These estimations correspond with the recent
382 plastic debris estimations for the Seine River. Additional studies should be performed on the
383 plastic debris flux variability in stormwater in other urban catchments, which could help in
384 more effectively estimating the plastic discharged into the environment.

385 6. Acknowledgements

386 We would like to thank the Urban Pollutants Observatory (OPUR) project for its support.

387 References

- 388 Althoff, J., Hebert, J., Grisoni, A., Châtel, L., Benattar, L., Buttin, G., 2020. Atlas du
389 plastique.
- 390 Barnes, D.K.A., 2002. Biodiversity: invasions by marine life on plastic debris. *Nature* 416,
391 808–809. <https://doi.org/10.1038/416808a>
- 392 Blettler, M.C.M., Abrial, E., Khan, F.R., Sivri, N., Espinola, L.A., 2018. Freshwater plastic
393 pollution: Recognizing research biases and identifying knowledge gaps. *Water Res.*
394 143, 416–424. <https://doi.org/10.1016/j.watres.2018.06.015>
- 395 Blettler, M.C.M., Ulla, M.A., Rabuffetti, A.P., Garello, N., 2017. Plastic pollution in freshwater
396 ecosystems: macro-, meso-, and microplastic debris in a floodplain lake. *Environ.*
397 *Monit. Assess.* 189, 581. <https://doi.org/10.1007/s10661-017-6305-8>
- 398 Chen, G., Feng, Q., Wang, J., 2020. Mini-review of microplastics in the atmosphere and their
399 risks to humans. *Sci. Total Environ.* 703, 135504.
400 <https://doi.org/10.1016/j.scitotenv.2019.135504>
- 401 Derraik, J.G.B., 2002. The pollution of the marine environment by plastic debris: a review.
402 *Mar. Pollut. Bull.* 44, 842–852. [https://doi.org/10.1016/S0025-326X\(02\)00220-5](https://doi.org/10.1016/S0025-326X(02)00220-5)
- 403 Dris, R., Gasperi, J., Saad, M., Mirande, C., Tassin, B., 2016. Synthetic fibers in atmospheric
404 fallout: A source of microplastics in the environment? *Mar. Pollut. Bull.* 104, 290–293.
405 <https://doi.org/10.1016/j.marpolbul.2016.01.006>
- 406 Dris, R., Gasperi, J., Tassin, B., 2018. Sources and Fate of Microplastics in Urban Areas: A
407 Focus on Paris Megacity. *Freshw. Microplastics* 69–83. [https://doi.org/10.1007/978-](https://doi.org/10.1007/978-3-319-61615-5_4)
408 [3-319-61615-5_4](https://doi.org/10.1007/978-3-319-61615-5_4)
- 409 Ellis, J.B., Butler, D., 2015. Surface water sewer misconnections in England and Wales:
410 Pollution sources and impacts. *Sci. Total Environ.* 526, 98–109.
411 <https://doi.org/10.1016/j.scitotenv.2015.04.042>
- 412 Gall, S.C., Thompson, R.C., 2015. The impact of debris on marine life. *Mar. Pollut. Bull.* 92,
413 170–179. <https://doi.org/10.1016/j.marpolbul.2014.12.041>
- 414 Gasperi, J., Dris, R., Bonin, T., Rocher, V., Tassin, B., 2014. Assessment of floating plastic
415 debris in surface water along the Seine River. *Environ. Pollut.* 195, 163–166.
416 <https://doi.org/10.1016/j.envpol.2014.09.001>
- 417 Gasperi, J., SEBASTIAN, C., Ruban, V., DELAMAIN, M., Percot, S., Wiest, L., Mirande, C.,
418 Caupos, E., Demare, D., DIALLO KESSOO, M., Saad, M., Schwartz, J., Dubois, P.,
419 Fratta, C., WOLFF, H., Moillon, R., Chebbo, G., Cren, C., MILLET, M., Barraud, S.,
420 Gromaire, M.-C., 2017. Contamination des eaux pluviales par les micropolluants:
421 avancées du projet INOGEV. *Tech. Sci. Méthodes* pp.51-66.
422 <https://doi.org/10.1051/tsm/201778051>
- 423 González-Fernández, D., Hanke, G., 2017. Toward a Harmonized Approach for Monitoring
424 of Riverine Floating Macro Litter Inputs to the Marine Environment. *Front. Mar. Sci.* 4.
425 <https://doi.org/10.3389/fmars.2017.00086>
- 426 Jambeck, J.R., Geyer, R., Wilcox, C., Siegler, T.R., Perryman, M., Andrady, A., Narayan, R.,
427 Law, K.L., 2015. Plastic waste inputs from land into the ocean. *Science* 347, 768–
428 771. <https://doi.org/10.1126/science.1260352>
- 429 Lebreton, L.C.M., Zwet, J. van der, Damsteeg, J.-W., Slat, B., Andrady, A., Reisser, J., 2017.
430 River plastic emissions to the world's oceans. *Nat. Commun.* 8, ncomms15611.
431 <https://doi.org/10.1038/ncomms15611>
- 432 Li, X., Chen, L., Mei, Q., Dong, B., Dai, X., Ding, G., Zeng, E.Y., 2018. Microplastics in
433 sewage sludge from the wastewater treatment plants in China. *Water Res.* 142, 75–
434 85. <https://doi.org/10.1016/j.watres.2018.05.034>

435 Liu, F., Olesen, K.B., Borregaard, A.R., Vollertsen, J., 2019. Microplastics in urban and
436 highway stormwater retention ponds. *Sci. Total Environ.* 671, 992–1000.
437 <https://doi.org/10.1016/j.scitotenv.2019.03.416>

438 Magni, S., Binelli, A., Pittura, L., Avio, C.G., Della Torre, C., Parenti, C.C., Gorbi, S., Regoli,
439 F., 2019. The fate of microplastics in an Italian Wastewater Treatment Plant. *Sci.*
440 *Total Environ.* 652, 602–610. <https://doi.org/10.1016/j.scitotenv.2018.10.269>

441 Mintenig, S.M., Int-Veen, I., Löder, M.G.J., Primpke, S., Gerdts, G., 2017. Identification of
442 microplastic in effluents of waste water treatment plants using focal plane array-
443 based micro-Fourier-transform infrared imaging. *Water Res.* 108, 365–372.
444 <https://doi.org/10.1016/j.watres.2016.11.015>

445 Mintenig, S.M., Löder, M.G.J., Primpke, S., Gerdts, G., 2019. Low numbers of microplastics
446 detected in drinking water from ground water sources. *Sci. Total Environ.* 648, 631–
447 635. <https://doi.org/10.1016/j.scitotenv.2018.08.178>

448 OSPAR Commission, 2010. Guideline for monitoring marine litter on the beaches in the
449 OSPAR maritime area.

450 Piñon-Colin, T. de J., al., 2020. Microplastics in stormwater runoff in a semiarid region,
451 Tijuana, Mexico. *Sci. Total Environ.* 704, 135411.
452 <https://doi.org/10.1016/j.scitotenv.2019.135411>

453 Pivokonsky, M., Cermakova, L., Novotna, K., Peer, P., Cajthaml, T., Janda, V., 2018.
454 Occurrence of microplastics in raw and treated drinking water. *Sci. Total Environ.*
455 643, 1644–1651. <https://doi.org/10.1016/j.scitotenv.2018.08.102>

456 Risch, E., Gasperi, J., Gromaire, M.-C., Chebbo, G., Azimi, S., Rocher, V., Roux, P.,
457 Rosenbaum, R.K., Sinfort, C., 2018. Impacts from urban water systems on receiving
458 waters – How to account for severe wet-weather events in LCA? *Water Res.* 128,
459 412–423. <https://doi.org/10.1016/j.watres.2017.10.039>

460 Schmidt, C., Krauth, T., Wagner, S., 2017. Export of Plastic Debris by Rivers into the Sea.
461 *Environ. Sci. Technol.* <https://doi.org/10.1021/acs.est.7b02368>

462 Schöneich-Argent, R.I., Dau, K., Freund, H., 2020. Wasting the North Sea? – A field-based
463 assessment of anthropogenic macrolitter loads and emission rates of three German
464 tributaries. *Environ. Pollut.* 263, 114367.
465 <https://doi.org/10.1016/j.envpol.2020.114367>

466 Talvitie, J., Heinonen, M., Pääkkönen, J.-P., Vahtera, E., Mikola, A., Setälä, O., Vahala, R.,
467 2015. Do wastewater treatment plants act as a potential point source of
468 microplastics? Preliminary study in the coastal Gulf of Finland, Baltic Sea. *Water Sci.*
469 *Technol.* 72, 1495–1504. <https://doi.org/10.2166/wst.2015.360>

470 Tramoy, R., Colasse, L., Gasperi, J., Tassin, B., 2019a. Plastic debris dataset on the Seine
471 river banks: Plastic pellets, unidentified plastic fragments and plastic sticks are the
472 Top 3 items in a historical accumulation of plastics. *Data Brief* 23, 103697.
473 <https://doi.org/10.1016/j.dib.2019.01.045>

474 Tramoy, R., Gasperi, J., Colasse, L., Noûs, C., Tassin, B., 2021. Transfer dynamic of
475 macroplastics in estuaries – New insights from the Seine estuary: Part 3. what fate
476 for macroplastics?

477 Tramoy, R., Gasperi, J., Dris, R., Colasse, L., Fisson, C., Sananes, S., Rocher, V., Tassin,
478 B., 2019b. Assessment of the Plastic Inputs From the Seine Basin to the Sea Using
479 Statistical and Field Approaches. *Front. Mar. Sci.* 6.
480 <https://doi.org/10.3389/fmars.2019.00151>

481 van Emmerik, T., Kieu-Le, T.-C., Loozen, M., Oeveren, K., Strady, E., Bui, X.-T., Egger, M.,
482 Gasperi, J., Lebreton, L., Nguyen, P.-D., Schwarz, A., Slat, B., Tassin, B., 2018. A
483 methodology to characterize riverine macroplastic emission into the ocean. *Front.*
484 *Mar. Sci.* 5. <https://doi.org/10.3389/fmars.2018.00372>

485 van Emmerik, T., Tramoy, R., van Calcar, C., Alligant, S., Treilles, R., Tassin, B., Gasperi, J.,
486 2019. Seine Plastic Debris Transport Tenfolded During Increased River Discharge.
487 *Front. Mar. Sci.* 6. <https://doi.org/10.3389/fmars.2019.00642>

488 van Sebille, E., Wilcox, C., Lebreton, L., Maximenko, N., Hardesty, B.D., van Franeker, J.A.,
489 Eriksen, M., Siegel, D., Galgani, F., Law, K.L., 2015. A global inventory of small

490
491
492

floating plastic debris. Environ. Res. Lett. 10, 124006. <https://doi.org/10.1088/1748-9326/10/12/124006>