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1 **Efficiency of source control systems for reducing runoff pollutant** 2 **loads: Feedback on experimental catchments within Paris conurbation**

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7 **Highlights:**

8 SUDS designed for peak flow control were monitored (flow, organic pollutants, metals)
9 Volume and contaminant loads were reduced by all SUDS even for ordinary events
10 Efficiency of SUDS depends strongly on the type of storage and its general conception
11 Strong reliance between pollutant mitigation and water volume reduction
12 Runoff reduction-oriented design of SUDS: an efficient solution for pollutant mitigation

14 **Abstract:**

15 Three catchments, equipped with sustainable urban drainage systems (SUDS: vegetated roof,
16 underground pipeline or tank, swale, grassed detention pond) for peak flow mitigation, have
17 been compared to a reference catchment drained by a conventional separate sewer system in
18 terms of hydraulic behaviour and discharged contaminant fluxes (organic matter, organic
19 micropollutants, metals). A runoff and contaminant emission model has been developed in
20 order to overcome land use differences. It has been demonstrated that the presence of peak
21 flow control systems induces flow attenuation even for frequent rain events and reduces water
22 discharges at a rate of about 50 % depending on the site characteristics. This research has also

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23 demonstrated that this type of SUDS contributes to a significant reduction of runoff pollutant
24 discharges, by 20 % to 80 %. This level of reduction varies depending on the considered
25 contaminant and on the design of the drainage system but is mostly correlated with the
26 decrease in runoff volume. It could be improved if the design of these SUDS focused not only
27 on the control of exceptional events but also targeted more explicitly the interception of
28 frequent rain events.

29 **Keywords:** flooding source control system; frequent rain event; micropollutants; stormwater;
30 SUDS

31

32 **1. Introduction**

33 Stormwater management has become a critical issue in the field of sustainable urban
34 development to protect civil society against flood and because runoff on urban surfaces has
35 been recognised as a major cause of the degradation of receiving waters (Burton and Pitt,
36 2001). In the past, stormwater was collected by drainage networks, but with fast urbanization
37 these networks have become inadequate, leading local authorities to develop strategies to
38 prevent flooding.

39 The first strategy adopted was the large-scale management of urban drainage systems by
40 building large reservoirs. It was not sufficient to remove the flooding risks and now a local
41 stormwater management approach is preferred (Brombach *et al.*, 2005; Ellis and Revitt, 2010;
42 Jefferies *et al.*, 2009; Roy *et al.*, 2008). In recently urbanised areas, facilities are developed
43 simultaneously to the urban growth promoting retention or infiltration at a small scale. These
44 facilities are often called “Sustainable Urban Drainage Systems” (or SUDS). Two major types
45 of SUDS design are used worldwide: flow rate regulation and volume regulation. Both in
46 France and in the USA, the most widespread regulation is based on a limited flow rate value
47 (Petrucci *et al.*, 2013; Roy *et al.*, 2008). For example in the French department Seine Saint-

48 Denis, in the suburb of Paris, the local authorities have imposed a flow rate regulation at
49 10 l/s/ha since 1993 (DEA, 1992). Thus SUDS are typically intended to facilitate hydraulic
50 management and have been designed for exceptional precipitation events; only on rare
51 occasion are contamination mitigation objectives actually addressed (Martin *et al.*, 2007).
52 Studies have revealed that such SUDS are capable of: reducing the discharged volumes,
53 delaying catchment response, slowing flow velocities and increasing water residence time
54 within the various facilities (Jefferies *et al.*, 2004; Scholes *et al.*, 2008). Thus they can have a
55 substantial impact on the pollutant fluxes being conveyed by stormwater and discharged into
56 receiving waters. Purifying effects have indeed been observed at the system scale for several
57 types of SUDS (Jefferies *et al.*, 2004; Pagotto *et al.*, 2000; VanWoert *et al.*, 2005). However,
58 there are few studies highlighting the overall effect of SUDS on pollutant fluxes control, at a
59 suburban catchment scale. The effect of SUDS that were designed for flow control and not
60 pollutant control remains poorly documented. Moreover literature data is usually limited to
61 metals and nutrients and few data is available on organic micropollutants (Dibiasi *et al.*, 2009;
62 Matamoros *et al.*, 2012).
63 Therefore, the objective of this research is to assess the effect of peak flow control policies,
64 on the water and contaminant flows discharged during frequent rain events at a small
65 catchment scale. A special attention has been given to a selection of priority substances listed
66 in the Water Framework Directive (2000/60/EC), whose presence is significant in runoff
67 (Bressy *et al.* 2012), but whose fate in SUDS is not much documented to date. Three
68 catchments containing SUDS were compared to a reference catchment featuring a
69 conventional separate sewer network, in terms of hydraulic behaviour and discharged
70 contaminant fluxes (i.e., suspended solids (SS), organic carbon (OC), trace metals (copper,
71 lead, zinc) and organic micropollutants: polycyclic aromatic hydrocarbons (PAHs),
72 polychlorinated biphenyls (PCBs), and alkylphenols). Moreover, the deposits formed in

73 storage zones were characterised so as to better understand the fate of micropollutants during
74 their transfer and in order to devise the best strategy for recovering and treating these wastes.

75

76 **2. Materials and methods**

77 *2.1 Site characterisation*

78 A residential site, characterised by low-density traffic and no industrial activity within a 5-km
79 radius, was studied in a suburban area near Paris (France). The site was drained by a separate
80 sewer system. Land use on this site was quite homogenous, while the stormwater
81 management system featured a wide diversity.

82 On this site, four small catchments ranging from 0.8 ha to 1.9 ha were studied. The
83 "Reference" catchment was drained by a conventional separate sewer system, while the other
84 three catchments ("North", "Park" and "South") temporarily stored stormwater in various
85 SUDS to comply with the 10 l/s/ha flow limitation imposed by local authorities. Stormwater
86 on the North catchment was stored in a vegetated roof and in an underground pipeline for
87 common rain events (up to 1 year return period) with an overflow onto a swale or on parking
88 for exceptional events. In the Park catchment, stormwater was stored in a grassed detention
89 pond that is part of a public garden. Stormwater management on the South catchment had
90 been incorporated into the land use plan and the practices associated various types of storage
91 facilities: underground tank for private parcels, swales and a public square covered by grass.
92 The outlets of the catchments with SUDS are fitted with flow-rate regulators as usual in
93 France (Table 1). According to Martin *et al.* (2007), these SUDS were representative of the
94 kinds of solutions adopted in France.

95 The characteristics of the catchments are listed in Table 1. The four catchments displayed a
96 homogeneous pattern of urbanisation and were located adjacent to one another (less than
97 400 m between two catchments), ensuring a relative homogeneity as regards atmospheric

98 contributions (i.e. rainfall and deposits). Differences in land use breakdown appeared across
99 these four catchments. The breakdown of the North catchment was similar to the Reference,
100 though automobile traffic was heavier on North due to the presence of retail shops. The Park
101 catchment was mainly composed of buildings and gardens and contains no streets. The South
102 catchment was relatively devoid of streets and contained a higher density of pedestrian paths
103 than the Reference. Consequently, the discharges for these four catchments could not be
104 directly compared and required introducing a set of land use-based modelling tools.

105 With the objective of establishing a model, the potential contaminant entry paths (atmospheric
106 fallout, pavement runoff and runoff from built parcels) were also examined. Atmospheric
107 fallout was measured on the flat roof of the highest building within the study area. Both types
108 of runoff (street and built parcel, i.e., roof and a private garden above slab underground
109 parking) were evaluated on the Reference catchment. The characteristics of these two sub-
110 catchments are provided in Table 1 and described with greater details in Bressy *et al.* (2011).

111

112 *2.2 Rainfall and flow measurements*

113 Rainfall and flows at the four catchment outlets were continuously measured (every 0.2 mm
114 for the rain and every minute for the flow) for one year between July 2008 and August 2009.

115

116 *2.2.1 Instrumentation*

117 Rainfall depth was recorded using a rain gauge (3029, Alcyr) placed on a flat roof in the study
118 area. Runoff flows were measured at the Reference catchment outlet with a Sigma 950 flow-
119 meter (water depth with a bubble pipe and velocity by Doppler Effect). At the source-
120 controlled catchment outlets, runoff flows were measured just beyond the flow regulation
121 device with Sigma 950 flow-meters, by recording the water depth upstream of a V-notch weir.

122

123 2.2.2 *Definition of a rain event*

124 A rain event was defined as any precipitation leading to a flow signal. The beginning of an
125 event was the time of the first precipitation data point during the 20 minutes preceding
126 initiation of the flow signal for the Reference site and during the preceding 3 hours for the
127 other three sites. The end of the event was defined as the time of the last precipitation data
128 point during the flow signal period.

129 The following parameters were determined for each rain event:

- 130 - Peak flow: Q_{\max} (in l/s/ha) was the maximum flow value during the rain event;
- 131 - Lag time: T_{lag} (h) was the time delay between maximum precipitation intensity and
132 peak flow signal;
- 133 - Emptying time: T_{empty} (h) was the time delay between the end of precipitation and the
134 end of the flow signal;
- 135 - Runoff water depth: H_{runoff} (mm) was the effective water depth discharged by the
136 catchment during the rain event.

137

138 2.3 *Sampling protocol and analytical procedure*

139 2.3.1 *Water sampling protocol*

140 Both dry and wet bulk atmospheric depositions were sampled using 20-L bottles hermetically
141 connected to a 1-m² stainless pyramidal funnel. The bottles were placed underneath the funnel
142 just before the rain event and removed just afterward; they collected the wet deposition and
143 the washoff of the contaminants deposited on the funnel during the previous dry weather
144 period. Stormwater was collected from the storm sewer at the catchment outlet using
145 automatic samplers (Bühler 1029) controlled via the flow meter. The sampling protocol was
146 flow-proportional so as to obtain average concentrations throughout the event.

147 The campaigns conducted in order to analyse both organic contaminants (requiring the use of
148 glass bottles) and metals (plastic bottles) were based on different sets of events. SS and
149 organic matter were measured for all the events in glass or plastic bottles, but in this paper
150 only SS and TOC data for rain events sampled simultaneously on atmospheric fallout, built
151 parcel and street catchments were used. Table 2 provides the characteristics of the rain events
152 considered for each parameter.

153

154 *2.3.2 Soil and sediment sampling protocol*

155 Sediment deposits were observed and sampled both in the North catchment storage pipe and
156 upstream of the South catchment regulator. Several samples were collected in order to
157 constitute a representative average sample of the sediment deposit. Average samples were
158 reduced by quartering steps after homogenization. The soil of the public garden used as
159 storage on Park catchment was also sampled. The retention basin surface was divided into
160 three areas according to flooding frequency: one flooded at each rain event, another
161 occasionally flooded and the last area was very seldom flooded. To constitute the average soil
162 sample, 4 to 6 samples were collected in each area using a (20-cm long) corer, in following a
163 7-m² mesh grid pattern, and then combined. One sample was analysed in triplicate for a
164 variability assessment: the signal deviation was below 20%.

165

166 *2.3.3 Micropollutant analysis*

167 The analytical procedures applied for organic compounds were previously described by
168 Bressy *et al.* (2012). Briefly, it was based on separating the dissolved and particulate fractions
169 (threshold: 0.45 µm). Dissolved fraction was extracted on a SPE C18 cartridge, while a
170 microwave-assisted extraction procedure was applied to the particulate fraction. The three
171 pollutant families (PCBs, PAHs and alkylphenols) were then separated during a purification

172 step on silica columns. Contaminants were quantified by internal calibration using gas
173 chromatography coupled with mass spectrometry (GC/MS, Focus DSQ, ThermoFisher
174 Scientific). Results are displayed as the sum of 13 PAHs³ (deriving from the US EPA list,
175 excluding naphthalene, acenaphthene and acenaphthylene, which are too volatile to be
176 correctly quantified), along with the sum of the 7 PCB indicators⁴. Among alkylphenols,
177 nonylphenols (NPs) and octylphenols (OPs) were studied.

178 Trace metals were analysed in both the total and dissolved fractions. Raw samples were
179 microwave acid-digested at 95°C with nitric and hydrochloric acids. Filtered samples on
180 0.45 µm cellulose acetate membranes were acidified to pH 1 with nitric acid. Metal
181 concentrations were determined using Inductively Coupled Plasma Atomic Emission
182 Spectroscopy (ICP-AES, Varian Vista MPX) through external calibration with a multi-
183 element standard solution (PlasmaNorm Multi-Elements).

184 The analytical uncertainties and the detection limits are given in Table S.1 in supplementary
185 files.

186

187 *2.4 Methodology used for site comparisons*

188 The various land use breakdowns did not exhibit the same runoff coefficient and moreover
189 did not produce the same pollutant quantities. Consequently a direct comparison of water and
190 contaminant fluxes between the Reference site and sites equipped with SUDS proved to be an
191 impossible task.

192 A water and contaminant emissions model was developed for each type of land use relative to
193 the Reference catchment. Then this model was applied to the land use breakdowns of each
194 catchment equipped with SUDS in order to simulate what the catchment behaviour would
195 have been in the absence of flow regulation. This methodology is shown Fig.1 and explained

³ Fluorene, phenanthrene, anthracene, fluoranthene, pyrene, benzo[a]anthracene, chrysene, benzo[b]fluoranthene, benzo[k]fluoranthene, benzo[a]pyrene, indeno[1,2,3]pyrene, di-benzo[a,h]anthracene, and benzo[g,h,i]perylene.

⁴ PCB 28, PCB 52, PCB 101 PCB 118, PCB 138, PCB 153, and PCB 180.

196 below. It is important to note that our methodology did not aim an accurate simulation of the
 197 pollutant masses but the estimation of SUDS effectiveness. For this purpose, the pollutant
 198 masses were simulated using low assumptions for sources, which allows to be sure that SUDS
 199 effect on pollutants is not overrated.

200

201 2.4.1 Tool for runoff volume simulation

202 The aim of this model was to simulate the runoff volume that would have been produced with
 203 conventional drainage systems based on the rainfall depth and land use dataset. The model
 204 was adapted from Berthier *et al.* (2001) and distinguished 3 types of surfaces: roofs, streets
 205 and gardens.

206 The initial losses on roofs and streets were modelled by a surface storage depth IL (mm). The
 207 filling level of this surface storage at the beginning of a rain event depended on the amount of
 208 rainfall during the previous 3 hours (H_{3h} in mm) for roofs and previous 6 hours (H_{6h} in mm)
 209 for streets. Infiltration through the street's pavement was modelled using a constant
 210 infiltration rate $K_{inf,s}$ (in mm/h). If H is the rainfall depth (mm), T_{rain} the rain event duration
 211 (h) and A_r and A_s the proportions of roof and street surface areas on the catchment, the runoff
 212 water depth (H_{runoff} in mm) was given by:

213 For roof runoff: $H_{runoff,r} = [H - \max\{IL_r - H_{3h}; 0\}]A_r$ **Equation 1**

214 For street runoff:

215 If $IL_s > H_{6h}$, then: $H_{runoff,s} = \max\{H - (IL_s - H_{6h}) - K_{inf,s} \cdot T_{rain}; 0\}A_s$ **Equation 2**

216 otherwise: $H_{runoff,s} = \max\{H - K_{inf,s} \cdot T_{rain}; 0\}A_s$

217 Private gardens above underground parking (50 to 100 cm soil) with drainage systems were
 218 modelled as a storage depth IL_g (mm), with their filling level at the beginning of the event
 219 being dependent on rainfall quantity over the previous 6 days (H_{6d} in h). The storage filling
 220 rate during the rain event was modelled by a constant infiltration rate $K_{inf,g}$;

221 evapotranspiration was not included in this modelling set-up. If A_g is the proportion of
 222 gardens covering the catchment, the runoff water depth from private gardens ($H_{runoff,g}$ in mm)
 223 was given by:

$$224 \quad \text{If } IL_g > H_{6d}, \text{ then: } H_{runoff,g} = \max\{H - \min\{IL_g - H_{6d}; K_{inf,g} \cdot T_{rain}\}; 0\} \cdot A_g \quad \text{Equation 3}$$

$$225 \quad \text{otherwise: } H_{runoff,g} = H \cdot A_g$$

226 The runoff water depth from the public garden was assumed to equal zero because, in a
 227 conventional system, it would not have been connected to the sewer system.

228 The five model parameters (IL_r , IL_s , IL_g , $K_{inf,s}$ and $K_{inf,g}$) were calibrated using rainfall and
 229 flow data over a 12-month period from the Reference catchment, by means of minimising the
 230 sum of absolute error values. The calibrated parameter values listed in Table 3 lie within the
 231 same interval as those found in the literature (Berthier *et al.*, 2001).

232

233 2.4.2 Tool for micropollutant simulation

234 The objective of this tool was to simulate contaminant emissions from catchments equipped
 235 with SUDS as if these catchments were being drained with a conventional sewerage system.

236 The principle consisted, for the sum of sampled events i (Cf. Table 2), of comparing the mass
 237 measured at the outlet (M) with the mass simulated by summing the masses input via the
 238 various entry paths (\bar{M}) according to the equations 4 and 5. The validation of the tool was
 239 done with the Reference data.

$$240 \quad M = \sum_i C_{Ref,i} \cdot V_i \quad \text{Equation 4}$$

$$241 \quad \begin{aligned} \bar{M} &= \bar{M}_{path} + \bar{M}_{street} + \bar{M}_{building} \\ &= \sum_i H_i \cdot \left[C_{atm,i} \cdot \overline{CR}_{s,i} \cdot S_p + C_{s,i} \cdot \overline{CR}_{s,i} \cdot S_s + C_{build,i} \cdot (\overline{CR}_{r,i} \cdot S_r + \overline{CR}_{g,i} \cdot S_g) \right] \end{aligned}$$

242 Where, for the event i , H_i is the rainfall depth; $C_{Ref,i}$, $C_{atm,i}$, $C_{s,i}$, $C_{build,i}$ the concentrations from Reference
 243 catchment outlet, atmospheric deposit, street runoff and building runoff according to Bressy *et al.* (2012);

244 $\overline{CR}_{s,i}$, $\overline{CR}_{r,i}$, $\overline{CR}_{g,i}$ the runoff coefficients calculated from the volumes simulated section 2.4.1 for street,
 245 roof and garden above underground parking; S_p , S_s , S_g the surfaces of path, street and garden; and V_i the
 246 water volume measured at Reference catchment outlet.

247 For zinc introduced via roofing materials, the corrosion models described in Gromaire *et al.*
 248 (2011) were used, distinguishing the zinc roofs of other:

$$\begin{aligned}
 \overline{M} &= \overline{M}_{path} + \overline{M}_{street} + \overline{M}_{building \neq zinc} + \overline{M}_{building = zinc} \\
 &= \sum_i \left[\begin{aligned} &H_{tot,i} \cdot C_{atm,i} \cdot \left(\overline{CR}_{s,i} \cdot S_p + \overline{CR}_{g,i} \cdot S_g + \overline{CR}_{r,i} \cdot S_{r \neq zinc} \right) \\ &+ H_{tot,i} \cdot C_{s,i} \cdot \overline{CR}_{s,i} \cdot S_s \\ &+ \overline{M}_{corrosion,i} \end{aligned} \right] \quad \text{Equation 5}
 \end{aligned}$$

250 Where, for the event i , $S_{r=zinc}$ and $S_{r \neq zinc}$ are the roof surfaces with zinc and without and $\overline{M}_{corrosion,i}$ the mass
 251 from corrosion (Gromaire *et al.*, 2011).

252 To avoid overestimating the simulated masses when data from Reference were missing due to
 253 technical problems, i.e., to avoid overvaluing the SUDS effect during comparisons with
 254 measurements, the simulated values were deliberately minimised by adopting hypotheses
 255 based on entry path concentrations and runoff volumes. Only atmospheric input was
 256 considered for PCB assuming the recent buildings or cars do not release them. For PAHs and
 257 alkylphenols, when concentrations from road runoff were missing, the lowest measured value
 258 was used. Uncertainties on model results induced by input data uncertainty (especially
 259 measurement uncertainty on experimental data) were estimated with the law of propagation of
 260 uncertainties (explanation in supplementary files Annexe A.1).

261

262 3. Results and discussion

263 3.1 Performance of the simulation tools

264 The errors and relative errors between simulated and measured H_{runoff} values are shown in
 265 Figure 2. Over a one-year period, model behaviour proved to be satisfactory (0.1% error
 266 between simulated and measured annual volumes) since the simulations have yielded good

267 results for the events producing the majority of yearly discharged water volume (60% of
268 events were simulated with a margin of error less than $\pm 30\%$, representing 80% of total
269 annual rainfall). To minimise errors, this model was always applied to the sum of studied
270 events, i.e., over the year for hydraulic simulation and over the sampled events for
271 micropollutant simulation.

272 Table 4 offers a comparison, for the Reference catchment, of measured mass vs. simulated
273 mass when aggregated over all sampled events (using Table 2 data). The simulated values lie
274 within the uncertainty of the corresponding measurement values (uncertainty on both water
275 volume and concentration measurements). This uncertainty does not cover scenario
276 uncertainty according to the classification described by Warmink *et al.* (2010) since the
277 scenario with the lowest value was initially chosen for sources. Our goal was not to develop a
278 model to simulate actual masses of pollutant but to assess the SUDS effect. This choice
279 allowed us to minimize the measured masses and therefore to avoid overestimation of the
280 SUDS efficiency.

281

282 *3.2 Effects of source control systems on discharged water*

283 *3.2.1 Flow dynamics at the event scale*

284 Flow dynamics were studied for all rain events between July 2008 and August 2009. Over this
285 period, 140 events could be distinguished on the Reference catchment (100 on North, 77 on
286 Park and 114 on South). The difference in number of events across catchments was due to the
287 slower dynamics of source-controlled catchments for which one event might correspond to
288 several for Reference. These results have been analysed with a focus on frequent rain events,
289 which represent most part of annual runoff volumes.

290

291

292 Peak flow reduction (see Fig. 3):

293 For Reference, Q_{\max} spanned a wide range of values (0.2 to 209 l/s/ha as the 1st and 9th
294 deciles), yet it remained below 10 l/s/ha for 69% of rain events suggesting that for the
295 majority of rain events the 10 l/s/ha regulatory flow threshold did not necessarily imply water
296 retention. The situation would be quite different with lower thresholds: for instance, the
297 2 l/s/ha level would be exceeded for 66% of events.

298 Figure 3 provides a peak flow reduction for all source-controlled catchments. This effect was
299 observed for almost all rain events, even ordinary rainfall episodes whose flow did not reach
300 the nominal regulator flow (10 l/s/ha). The flow rate actually exceeded 2 l/s/ha for 19% and
301 13% of events at the North and Park outlets. At the South outlet, flow exceeded 2 l/s/ha for
302 63% of events; consequently, source control systems in place at the South site caused less
303 impact on frequent rain events.

304 In the case of North catchment, peak flows were mainly controlled by the flow regulator.
305 However, for Park and South catchments, the level required for the initiation of the regulator
306 was not reached for most rain event and peak flow attenuation was due to the natural retention
307 time in the garden and in the swales.

308

309 Staggering and lag time of discharged water (see Fig.4):

310 For the Reference catchment, flow dynamics were close to the rainfall dynamics: T_{lag} varied
311 between 3 and 14 minutes (1st - 9th deciles), and T_{empty} ranged from 0.8 to 5.8 hours. These
312 high T_{empty} values were due to drainage in the private gardens above underground parking.

313 For source-controlled catchments, the peak flows shifted in time relative to the rain peak: T_{lag}
314 varied from 16 minutes to 2.3 hours for North, 23 minutes to 2.6 hours for Park, and 8
315 minutes to 1.4 hours for South (1st - 9th deciles).

316 The North and Park catchments also showed much longer emptying times for the aggregate of
317 all rain events than either Reference or South. Storage emptying lasted between 1.7 and 17
318 hours for North and 2.3 to 10 hours for Park. The South catchment took between 0.9 and 3
319 hours to empty, which was of the same order of magnitude as Reference. The reactivity of the
320 South site could be explained by the type of regulation system installed (vortex regulator,
321 pump) with which flows quickly reached the nominal regulator flow. It may induce fewer
322 effects when a small quantity of water is being stored, i.e., for ordinary rain.

323 These results indicate that the design of retention devices and, more importantly, the choice of
324 regulation system have proven to be determinant as regards flow dynamics.

325

326 *3.2.2 Effect on discharged volumes*

327 For the three source-controlled catchments, the annual discharged volumes that would be
328 generated with a conventional storm sewerage system were simulated with the model
329 described in the section 2.4.1 and then compared in Table 5 to actual measured volumes as
330 regards annual runoff coefficient (calculated as the ratio of rainfall amount to runoff water
331 depth).

332 The annual runoff coefficient simulated for a conventional sewerage system was very close to
333 the proportion of impervious surfaces for all 4 catchments. A large reduction (43% to 55%) in
334 annual runoff volumes, compared to the hypothetical volumes with a conventional storm
335 sewer, was observed for all 3 source-controlled catchments. This significant reduction was not
336 surprising for the Park catchment since all its stormwater flowed through a garden and
337 potentially infiltrated into the soil. For South, the reduction was correlated with longer and
338 more extensive contact with vegetated surfaces (grass strips, swales, grassed basin). The
339 reduction was much less expected on the North site, where storage facilities are mostly
340 composed of impervious materials. One part of this reduction could be explained by vegetated

341 roof and the other part by greater initial losses within the source control systems (dead
342 volumes).

343 These reductions in annual runoff volumes represent very promising developments for
344 stormwater quality management. Discharged masses are indeed partly determined by
345 discharged volumes, as already observed by other authors (Davis *et al.*, 2009; Trowsdale *et*
346 *al.*, 2011), and lower annual runoff volumes can also induce a reduction in the discharged
347 pollutant load.

348

349 *3.3 Effects of source control systems on pollutant loads in stormwater*

350 Figures 5a, 5c and 5e display, for all contaminants studied, both the masses measured (grey
351 histogram) at the outfall of each catchment and the simulated masses (white histogram) as if
352 the storm drainage system was conventional, for the sum of all monitored events. Water
353 volume data have been added for comparison. Figures 5b, 5d and 5f present the same types of
354 data for average concentrations. Uncertainty bars associated with measured data correspond to
355 measurement uncertainties and those associated with simulated data are model uncertainties
356 (with a 80 % confidence interval) (explanation in supplementary files Annexe A.1).

357

358 *3.3.1 North catchment*

359 For the North catchment, the mass values simulated for a situation with no SUDS ever
360 installed were higher than the measured masses on the whole, a finding that reveals a drop in
361 onsite contamination. The effect on contaminant concentrations depended on the type of
362 substance. Three categories of behaviour could be distinguished:

363 - For SS, total PAHs and zinc, the decrease in contaminant mass (50%, 60% and 72%,
364 respectively) was greater than the drop in water volume (43%). The measured concentrations
365 were thus lower than the simulation results. Since SS and PAHs are both particulate, we

366 assumed the level of settling would be substantial in the underground storage zones, given
367 that these decreases amounted to the same order of magnitude as the mass reductions
368 evaluated for large storage basins (Aires *et al.*, 2003; Calabro and Viviani, 2006; Clark and
369 Pitt, 2012). The introduction of zinc, on the North site, occurred mainly in dissolved form via
370 the corrosion of metallic roof materials (80% in dissolved form according to Bressy *et al.*
371 (2012)). At the outfalls of large catchments, zinc has been proved to be 50% bind to
372 particulate matter (Zgheib *et al.*, 2011), which proves that zinc tends to bond with particles. It
373 is suggested that part of the dissolved zinc became attached to particles and settled with them
374 or else bonded with either the drainage system or deposits in the storage zone.

375 - For PCBs and NPs, in their total and dissolved form, the decrease in contaminant mass
376 (between 24% and 36%) was assumed equal to the drop in water volume since the simulated
377 concentrations lied within the uncertainty interval of measured concentrations. These mass
378 amounts were thus lowered by the presence of storage, yet at a constant concentration. Let's
379 recall that these substances were at around 70% in dissolved form in our samples (Bressy *et*
380 *al.*, 2012). This speciation was not expected to differ like it did for zinc since the distribution
381 here is the one measured in stormwater (Zgheib *et al.* 2011) or for the natural environment
382 (Cailleaud *et al.*, 2007). It is therefore likely that a portion of the contamination has been
383 trapped during water losses due to sedimentation / filtration for the particulate fraction and to
384 adsorption / infiltration for the dissolved fraction.

385 - For the total (TOC) and dissolved organic (DOC) carbon, copper, dissolved PAHs and both
386 total and dissolved OPs, the decrease in contaminant mass when assuming no SUDS had been
387 installed was less than the drop in water volume or lied within the uncertainty interval of the
388 measures. The mass of released contaminants was in fact lower by use of on-site storage,
389 although the concentration released was slightly superior to that simulated for conventional
390 sewer. These substances are in the both fractions (Bressy *et al.*, 2012) and should therefore

391 undergo at least the same decrease as the other substances, i.e. by sedimentation for the
392 particulate fraction and adsorption for the dissolved fraction. One hypothesis for these
393 findings might be that our simulation has underestimated the masses of these substances, as
394 automotive traffic is a major source of copper, PAHs and OPs (Bjorklund *et al.*, 2009; Bressy
395 *et al.*, 2012; Motelay-Massei *et al.*, 2006). Automotive traffic is more intense on the North
396 catchment than the Reference site, which was used to calibrate the street-based contaminant
397 production function. But as explained in paragraph 3.1, our model intentionally
398 underestimates the simulated masses in order to avoid overestimation of the SUDS effect. As
399 a consequence, for these substances, our methodology does not allow us to conclude about the
400 effectiveness of SUDS.

401

402 3.3.2 Park catchment

403 Downstream of the Park catchment, the majority of simulated mass values exceeded the
404 measured values (except for SS and PAHs in the total fraction), thus indicating a mass drop
405 due to the use of open space storage. As mentioned for the North catchment, this decreasing
406 effect depended on the type of substance under consideration:

407 - For zinc, dissolved PAHs, total and dissolved NPs, and total and dissolved OPs, the decrease
408 in contaminant mass (80%, 71%, 70% and 60%, respectively) was greater than or equal to the
409 loss of water volume (60%). For these substances, which are mainly in dissolved form (80%
410 for zinc, 79% for NP and 74% for OP according to Bressy *et al.* (2012)), an adsorption effect
411 in the public garden was to be assumed, as demonstrated by Ray *et al.* (2006) on tree bark
412 samples or by Scholes *et al.* (2008).

413 - For TOC, DOC, PCBs, dissolved PCBs and copper, the contaminant mass decrease was less
414 than the actual volume loss. The simulated mass exceeded measurement results by
415 respectively 29%, 29%, 42%, 50% and 22%, with measured concentrations topping the

416 simulated concentration values. It was possible that the adsorption effect for these substances,
417 which were at 57% in particulate fraction for TOC, 36% for PCBs and 72% for copper
418 (Bressy *et al.*, 2012), was less pronounced than for the group of substances described above.

419 - The mass of SS and total PAHs did not appear to be reduced by the upstream management
420 systems introduced. The measured masses of these substances exceeded the simulated values,
421 as if the fact of regulating discharges were raising the level of water contamination for these
422 parameters. It was likely that our simulation has underestimated the actual SS mass produced,
423 given that this value did not take into account particle production from the garden and
424 playground (sandpit). As above, for these substances, our methodology does not allow us to
425 conclude about the SUDS efficiency.

426 On this site, the decrease in water volume was high (over 50%), and the water residence time
427 in the basins was quite long according to Section 3.2.1 (median drainage time exceeds 5
428 hours). The contact time between water and potential substrates (plants and soil particles) was
429 also increased, thereby promoting both the adsorption of dissolved contaminants and their
430 infiltration into the soil. Moreover, since the storage facilities were not enclosed, it was
431 considered likely that the phenomenon of volatilisation, photolysis and biodegradation
432 eliminated a portion of soil contamination during dry weather periods (Scholes *et al.*, 2008;
433 Weiss *et al.*, 2007).

434

435 3.3.3 South catchment

436 On the South catchment, all simulated mass values exceeded measurements, which definitely
437 points to contaminant interception within the various SUDS systems. This decrease in mass
438 release however remains small in magnitude given that all simulated concentrations was less
439 than or equal to the measured concentrations, with the exception of zinc.

440 - As regards zinc, the mass decrease (60%) exceeded the drop in water volume (46%), and the
441 simulated zinc concentration was 24% higher than the measured value. On this catchment, the
442 majority of zinc (90%) entered in dissolved form through the corrosion of roofing materials
443 (Gromaire *et al.*, 2011) and the roof runoff was recovered in both an underground tank and
444 planted swales. In these storage zones, sorption may indeed occur.

445 - As regards DOC, dissolved PCBs, dissolved PAHs, total and dissolved NPs, total and
446 dissolved OPs and copper, the mass decline was similar to the reduction in water volume.
447 Since all these parameters were essentially dissolved (Bressy *et al.*, 2012), sorption or
448 infiltration effects were clearly apparent.

449 - The decreases in SS mass (23%), TOC (33%), total PCBs (15%) and total PAHs (19%) were
450 all less than the loss of water volume (46%): these parameters tend to be more particulate in
451 nature.

452 The decrease in South catchment contaminant mass thus appeared to be less pronounced than
453 that of the other sites. Flow control systems on this catchment were less efficient on current
454 flow rates than the other studied sites (Section 3.2.1), which results in lower residence times
455 and hence worse efficiency.

456

457 *3.3.4 Fate of micropollutants in these systems*

458 Level of deposit contamination:

459 Figure 6 presents the contaminant contents in deposits at both the North (underground pipe)
460 and South (storage zone upstream of the regulator) catchments.

461 The average PCB contents varied between 0.034 $\mu\text{g/g.dw}$ for South and 0.058 $\mu\text{g/g.dw}$ for
462 North: this range was 3 times weaker than contents measured in the Reference stormwater SS
463 (0.10 $\mu\text{g/g.dw}$, according to Bressy *et al.* (2012)). The particles held in storage prove to be the
464 coarsest as well as the least contaminated; in addition, they are comparable to those detected

465 by Jartun *et al.* (2008) in sediments from a separate urban sewer system in Norway (0.029
466 $\mu\text{g/g.dw}$) and below the limit established by the French decree relative to the spreading of
467 sewage sludge (Decree No. 97-1133, 1998), i.e. $0.8 \mu\text{g/g.dw}$.

468 Average PAH contents was equal to $6.9 \mu\text{g/g.dw}$ in South sediment and $7.0 \mu\text{g/g.dw}$ in North
469 sediment, which places them at roughly 5 times less than the contents measured in the
470 Reference SS ($33 \mu\text{g/g.dw}$). These results were comparable to the values measured by Gasperi
471 *et al.* (2005) in particles from water used for street cleaning and above the contents recorded
472 by Jartun *et al.* (2008): $3.4 \mu\text{g/g.dw}$. The contents were between 3.5 and 4 times weaker than
473 the limits established for the spreading of sewage sludge.

474 NPs contents ranged from $0.26 \mu\text{g/g.dw}$ at South to $6.3 \mu\text{g/g.dw}$ at North (0.04 and
475 $0.70 \mu\text{g/g.dw}$ for OPs, respectively). The differences identified between these two sites may
476 be explained by a smaller proportion of road on South and a building age effect subsequent to
477 the European NP use restriction Directive (Directive 2003/53/EC). More specifically, the
478 South catchment contained more recent construction, meaning that the materials employed
479 could contain less NP. The contents found in Reference SS were measured at $6.8 \mu\text{g/g.dw}$
480 ($0.27 \mu\text{g/g.dw}$ for OPs). Bjorklund *et al.* (2009) found lower contents in sediments from a
481 separate sewer network in Sweden: below $1.5 \mu\text{g/g.dw}$ for NPs, and below detection limits for
482 OPs. No regulation has been adopted for alkylphenols as regards their spreading.

483 For trace metals, the contents recorded at South revealed: 0.059 mg/g.dw copper, 0.041
484 mg/g.dw lead, and 2.8 mg/g.dw zinc. North catchment results yielded: 0.16 mg/g.dw copper,
485 0.12 mg/g.dw lead, and 0.79 mg/g.dw zinc. The higher copper and lead contents found in
486 North stemmed from the greater volume of road traffic. The lower zinc values in North were
487 due to a much larger proportion of zinc roofs in South (Table 1). These entire values were
488 lower than those for Reference particles: 0.28 mg/g.dw copper, 0.26 mg/g.dw lead, and
489 5.5 mg/g.dw zinc. The order of magnitude remained the same as for measurements conducted

490 by Jartun *et al.* (2008), i.e., 6 times less than the spreading limits for copper, 7 times less than
491 those for lead and 4 times for zinc.

492 These deposits were thus only slightly contaminated, especially when compared to the
493 concentrations measured in stormwater SS: the coarsest, and hence least contaminated,
494 particles are those retained in the SUDS. This category of particles does not require any
495 special treatment, as opposed to the sludge generated by combined sewer networks.

496

497 Impacts of urban runoff retention on soil contamination:

498 To understand the fate of contaminants and to assess whether stormwater storage actually
499 degrades garden soil quality in Park catchment or not, contaminant contents were measured in
500 the soil as a function of the flooding gradient. Figure 7 displays the contents recorded across
501 the various zones. OPs are not displayed because contents were around the LOQ.

502 PAH concentrations varied between 0.33 and 1.1 $\mu\text{g/g.dw}$, while PCB contents ranged from
503 0.026 to 0.060 $\mu\text{g/g.dw}$ and NP values from 0.22 to 0.45 $\mu\text{g/g.dw}$. These PAH and PCB
504 contents are comparable to measurements recorded in Seine River basin soils by Motelay-
505 Massei *et al.* (2004) and are below the values from the Canadian Soil Quality Guidelines
506 (CCME, 2007) and far from the intervention values for remediation given by the regulation of
507 Netherlands (VROM, 2000). In contrast, the garden soil seems to be contaminated by NPs,
508 given the findings of Vikelsoe *et al.* (2002), who reported contents equal to 0.034 $\mu\text{g/g.dw}$ in
509 runoff storage areas, but this value stay below the Canadian Soil Quality Guidelines (CCME,
510 2007).

511 Copper contents varied between 0.014 and 0.040 mg/g.dw , lead between 0.020 and
512 0.083 mg/g.dw , and zinc between 0.056 and 0.10 mg/g.dw . These values are of the same
513 order of magnitude as Paris region soils, according to Thévenot *et al.* (2007) and they respect
514 guidelines (CCME, 2007; VROM, 2000).

515 A comparison across the 3 zones of increasing flood frequency did not reveal any difference
516 in content for PCBs, and NPs. For PAHs and 3 trace metals, however, the most commonly
517 flooded zones showed contents of between 1.3 and 3.2 times less. These differences may be
518 attributed to soil heterogeneity or to an infiltration or degradation process, depending on
519 contaminants. Moreover, these results demonstrate that the Park catchment soil is not
520 significantly contaminated following its use for storage.

521

522 *3.4 Discussion on micropollutant reduction measures for stormwater*

523 Section 3.2 highlighted: i) a significant reduction in annual runoff volumes at the outlets of
524 the source-controlled catchments due to losses on permeable surfaces and higher initial losses
525 on impervious surfaces and ii) an increase in water retention time, which strongly depend on
526 facilities design. When the design of SUDS considers only exceptional rainfalls, retention
527 efficiency of common rainfall is highly variable depending on the type of storage (permeable
528 or impervious surfaces) and the method of flow regulation. For the facilities that naturally
529 increase the retention time, like the large grassed detention garden in Park catchment, the
530 effects due to the flow regulator itself become insignificant for common rainfall.

531

532 Section 3.3 proved that the presence of SUDS considerably lowered the contaminant masses
533 released by the catchments. This conclusion was based on a comparison of fluxes measured
534 with SUDS and simulated without SUDS. This methodology takes into account the
535 measurement uncertainty. Scenario uncertainty was not evaluated, however the simulated
536 scenario is an underestimating scenario. It is intentionally based on minimal emissions from
537 the different pollutant sources, so as to avoid overestimation of the SUDS effectiveness.

538 This decrease varied from 20% to 80% of the total released mass, depending on both the
539 study site and the type of pollutant. For underground storage facilities, the drop is substantial

540 for particulate pollutants (SS, particle-bound contaminants) which are sensitive to
541 sedimentation, and less pronounced for dissolved contaminants which might adsorb to the
542 structures or deposits. For storage scenarios in permeable planted zones, the most significant
543 contaminant decreases involve dissolved pollutants. Other effect of the SUDS can be
544 observed on the substances primarily introduced into stormwater in dissolved form (i.e.
545 metals from the corrosion of roofing materials) and that exhibits a strong affinity for organic
546 matter. In this case, the binding of these substances to deposits and soil is assumed, inducing a
547 stronger reduction of their release. Our findings should apply to other sites equipped with
548 SUDS if they are designed with the same criteria. These results are interesting at local scale
549 because it provides original data on organic micropollutants in SUDS and particularly useful
550 at global scale because they allow to make effective recommendations for design criteria in
551 terms of reducing pollution.

552

553 The effects on discharged contaminants may be correlated with hydraulic effects, namely the
554 followings.

555 - Discharged masses are strongly correlated to discharged volumes and a reduction in
556 annual runoff volumes may explain the highlighted mass decreases. Consequently, the
557 volume drop for ordinary rain events needs to be assigned a top priority when developing
558 stormwater management processes and, moreover, can be maximised by adding storages
559 in grassy areas.

560 - During runoff, the reduction in flow velocity may lower contaminant erosion at the
561 surface.

562 - In water storage facilities, reduced flow velocity allows for particle settlement (Calabro
563 and Viviani, 2006). A longer residence time increases contact time between water and
564 substrate (i.e., soil, sediments, building materials) and is therefore favourable to both

565 pollutant filtration through permeable materials and sorption of dissolved fractions (Mason
566 *et al.*, 1999; Ray *et al.*, 2006).

567 - On a long-term basis and during dry weather periods, some of the pollutants retained in
568 the source control devices may undergo evaporation or degradation (biodegradation,
569 photodegradation, etc.) (Scholes *et al.*, 2008; Warren *et al.*, 2003).

570 In order to improve water quality, the design of SUDS should focus on systems that retain all
571 rain events, even those with the lowest intensity. To intercept ordinary events without
572 increasing the size of storage facilities, flow rate regulation may be adjusted according to the
573 importance of the event. It may be feasible to store the first few millimetres of rain in porous
574 materials without discharge and by emptying via infiltration and/or evapotranspiration. As an
575 example, "rain garden" and "bioretention" type systems are currently being promoted in other
576 countries (Dietz and Clausen, 2005; Jefferies *et al.*, 2004). These recommendations are
577 consistent with Petrucci *et al.* (2013). They showed that flow-rate based regulations can
578 produce negative impacts on water discharge at the catchment scale and that volume-based
579 regulations should be encouraged for example local infiltration facilities.

580

581 **4. Conclusion**

582 The research presented herein has demonstrated that the use of SUDS systems, initially aimed
583 at peak flow attenuation, can also serve to slow and delay water flow for frequent rain events,
584 and result in a significant reduction of the annual discharged volumes. The masses of
585 discharged contaminants are also decreased. This phenomenon is correlated with hydraulic
586 effects: greater initial and continuous losses limit contaminant transfer downstream while an
587 extended residence time enhances the phenomena of substance sedimentation and adsorption.
588 This reduction in discharged contaminants is primarily explained by a drop in runoff water
589 volumes. These results do not systematically reveal any kind of "purifying effect" in the

590 classical meaning (i.e. lower concentrations), but instead an overall effect of reducing mass
591 discharges.

592 The effects are however highly variable from one site to another, and from one contaminant to
593 another. They depend on the hydraulic interception of small rainfalls. For impervious storage
594 structures the retention time of small events depends mainly on the characteristics of the flow
595 regulator device, whereas for pervious and vegetated storage structures natural losses
596 (infiltration, evapotranspiration) greatly contribute to the interception of frequent rain events.

597

598 Bressy *et al.* (2012) have exposed the benefit of managing stormwater upstream, for the
599 purpose of locally discharging the slightly contaminated water; refraining from mixing with
600 heavily polluted water, and avoiding the network cross contamination process.

601 Nowadays, in France, the design of SUDS is mainly intended to protect against flooding and
602 to limit discharged flows mainly by intercepting the exceptional rain events. To ensure
603 efficiency in terms of pollutant mitigation, this upstream stormwater management approach
604 must limit water transfer downstream and take into account frequent rain events when
605 designed. A regulation system needs to be introduced for retaining ordinary events without
606 excessively increasing the storage volume for ten-year return period rainfalls. As an example,
607 the first few millimetres of rainfall could be systematically stored in a porous material or a
608 vegetated zone without any discharge to the network but with drainage provided by
609 infiltration and/or evapotranspiration.

610

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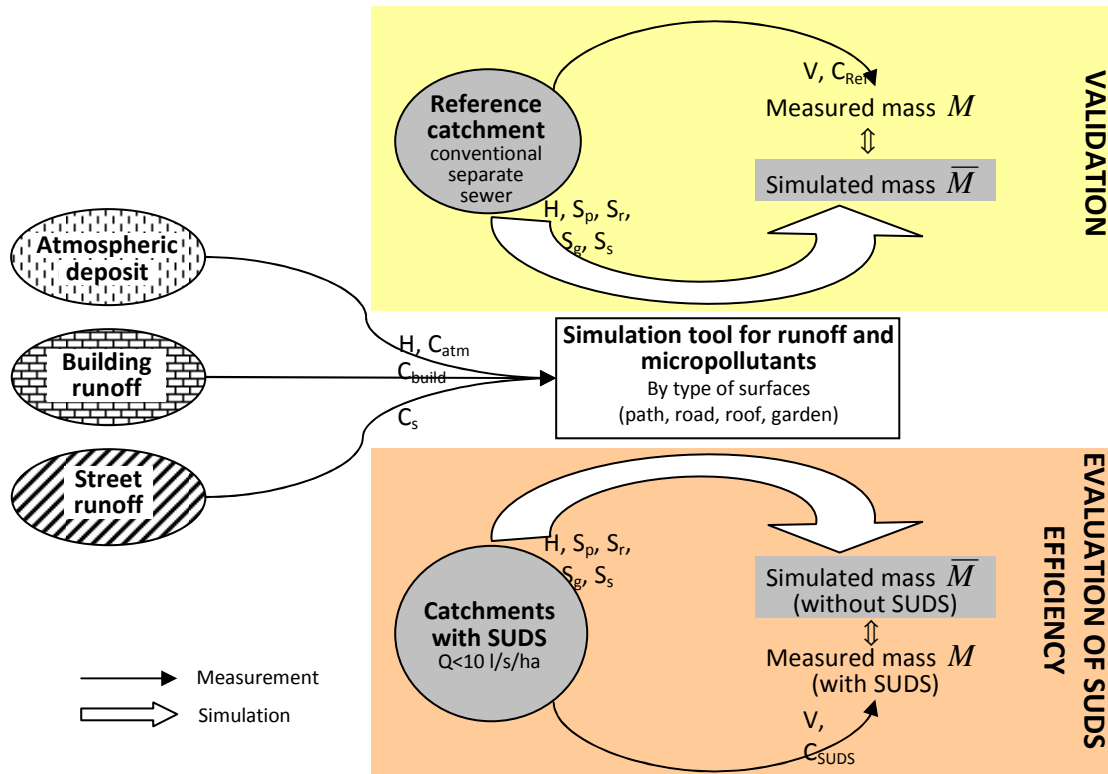
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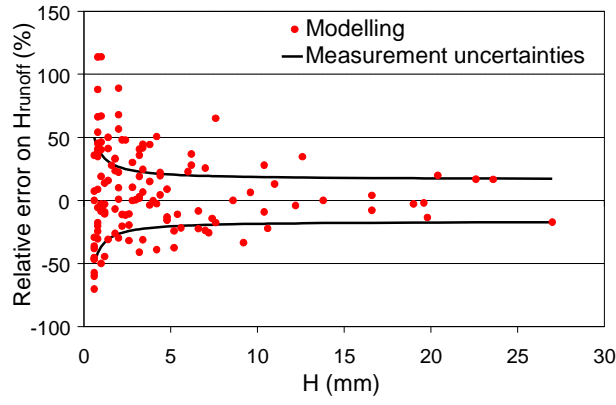


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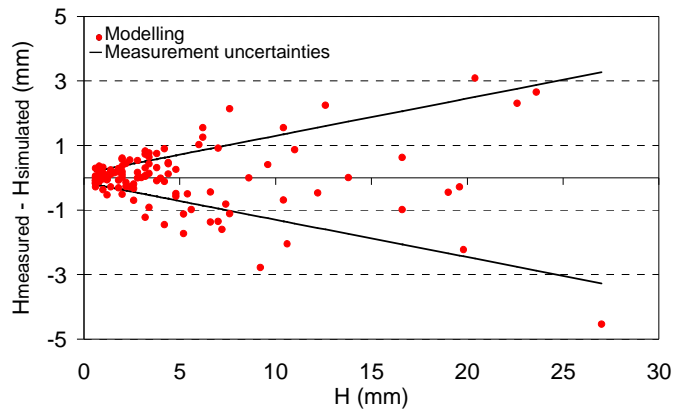
2 **Fig. 1.** Methodology for comparison between sites (with H the rainfall depth, V the runoff

3 volume, S the surface, C the concentration, Q the flow rate and M the mass)

1



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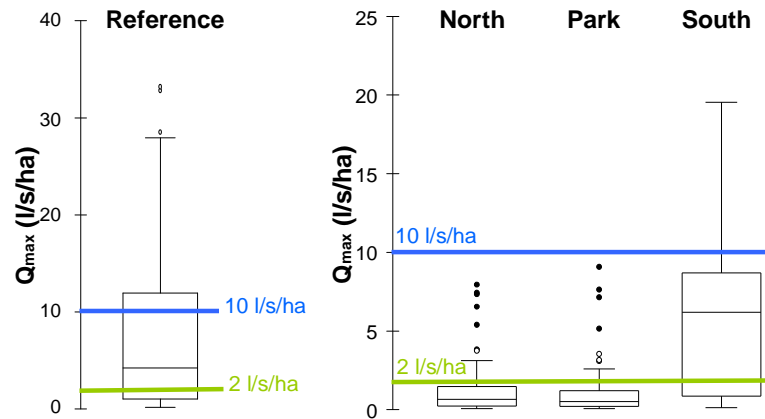


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Fig. 2. Errors and relative errors of the model vs. rainfall depth

4

1

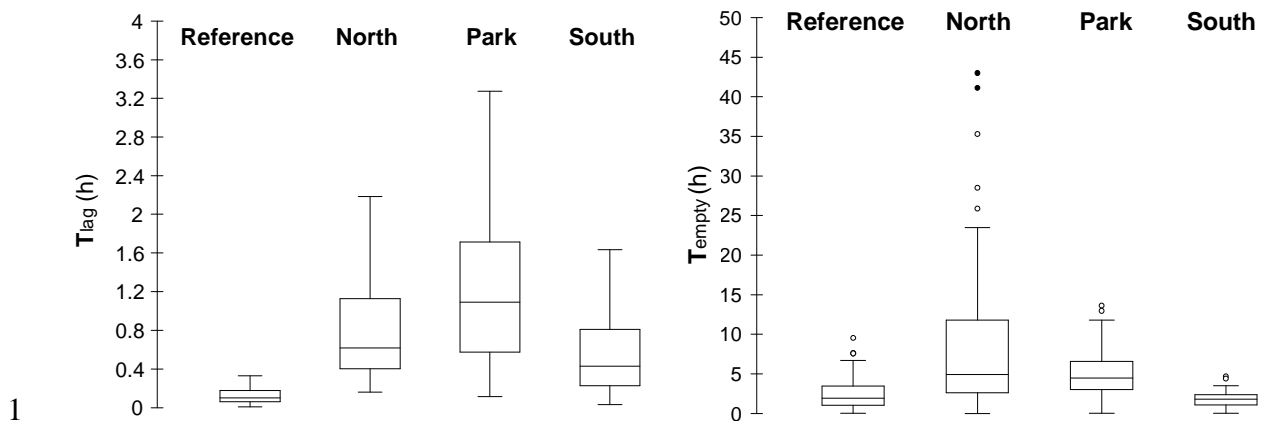


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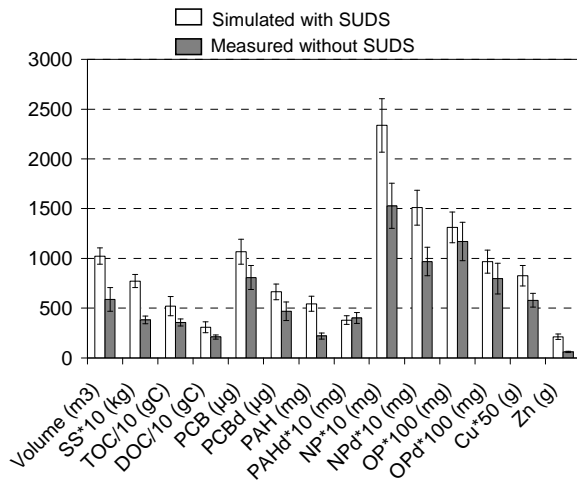
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Fig. 3. Box plots⁵ of peak flows observed during events from a one-year monitoring period

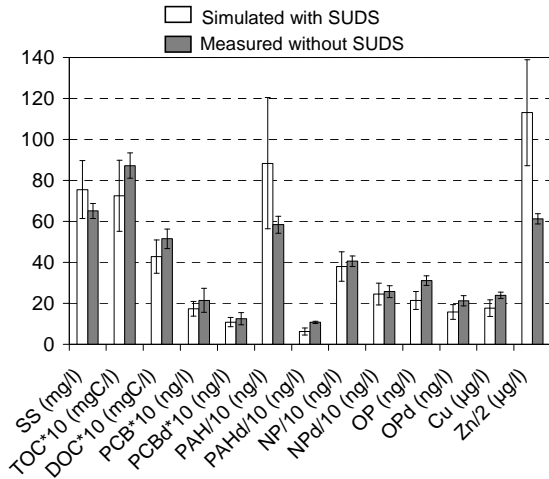
⁵ The box represents the 25th and 75th percentiles and the band inside represents the median. The ends of the whiskers depict the lowest data remaining within the 1.5 interquartile range (IQR) of the lower quartile and the highest data remaining within 1.5 IQR of the upper quartile.



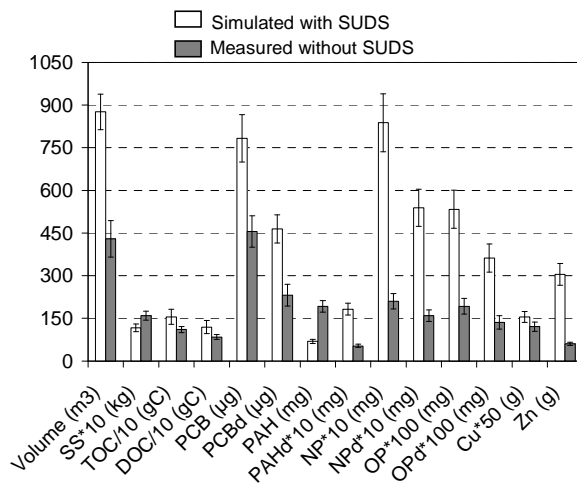
1
2 **Fig. 4.** Box plots of T_{lag} and T_{empty} for each catchment and all rain events between July 2008
3 and August 2009



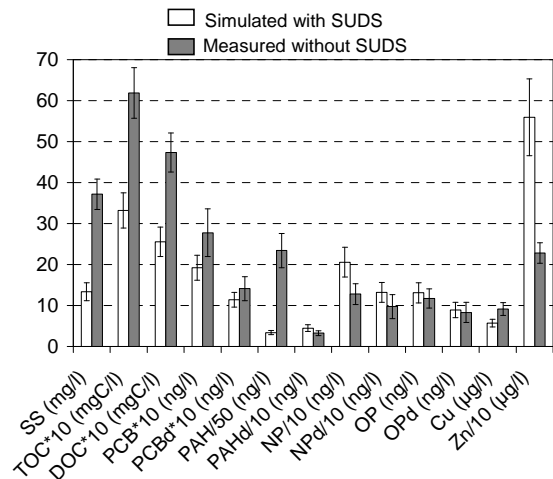
5a. Contaminant masses on North



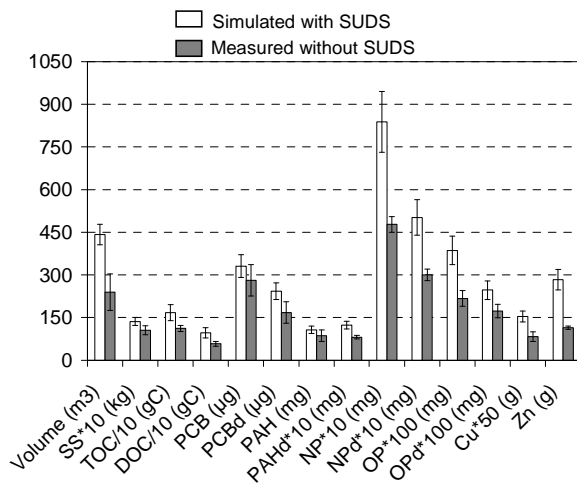
5b. Contaminant concentrations on North



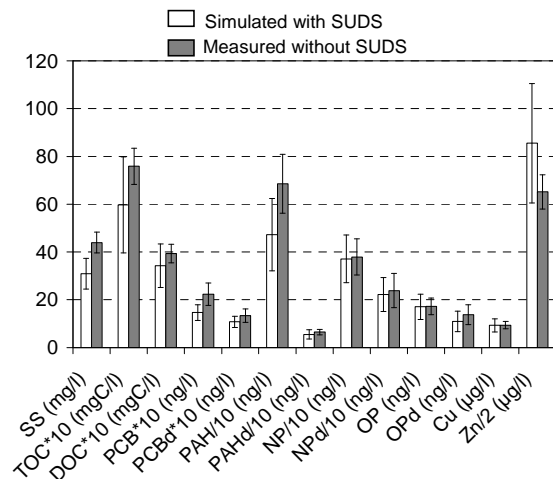
5c. Contaminant masses on Park



5d. Contaminant concentrations on Park



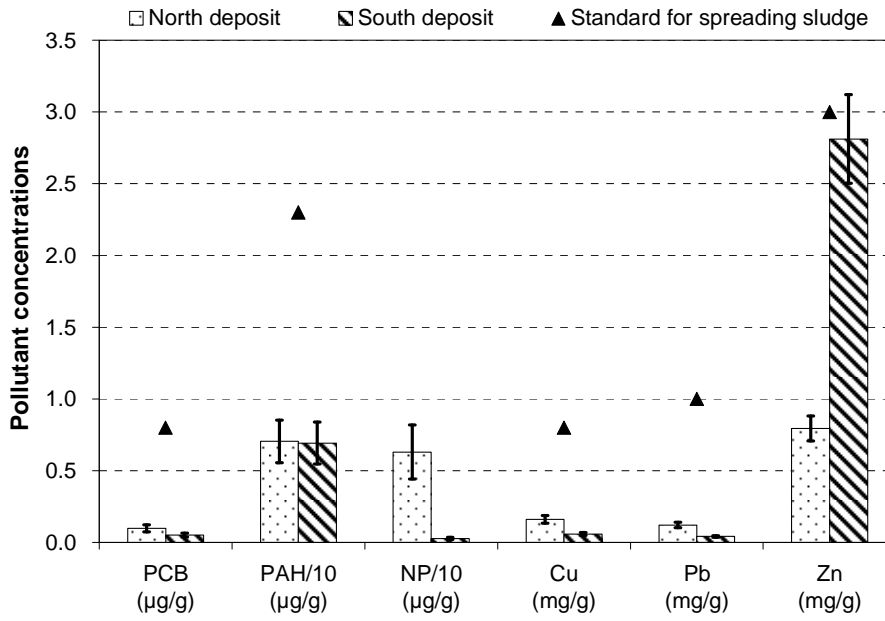
5e. Contaminant masses on South



5f. Contaminant concentrations on South

Fig. 5. Masses and concentrations simulated without SUDS and measured at the catchment outlet for the sum of monitored events

1

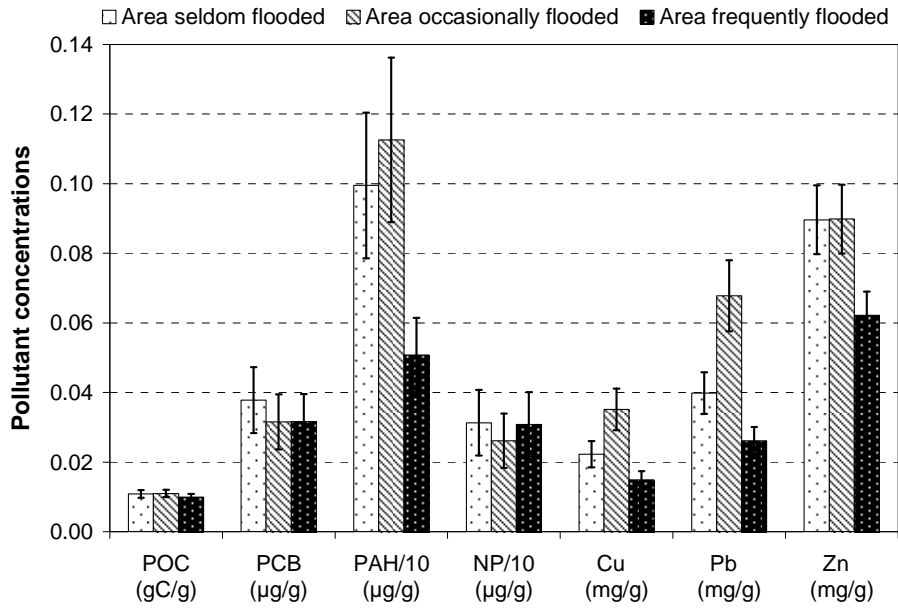


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Fig. 6. Pollutant concentrations (in dry weight) in the North and South catchment deposits

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Fig. 7. Pollutant concentrations in the soil of Park catchment retention basins

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Table 1. Description of the studied catchments

Name	Size (ha)	Land use (%)						Retention system	Flow regulation
		R≠Zn*	R=Zn*	S*	P*	Gs*	G*		
Reference	0.82	36	7	28	3	25	0	Conventional stormwater system	-
North	1.5	47	2	24	4	18	6	Vegetated roof + retention in an oversized pipe (return period of up to 2 years), possible overflow into a swale and a parking lot	16 l/s vortex flow regulator
Park	2.0	12	4	0	19	26	39	2 grassed retention basins in a public park	23 l/s float valve flow regulator
South	0.92	28	10	8	19	17	17	Swales + grassed retention basin in a square + underground tank	2 × 1 l/s nozzles + 5.6 l/s vortex flow regulator + 3 l/s pump
Built parcel	0.13	22	43	0	4	31	0	Conventional stormwater system	-
Street	0.031			100				Conventional stormwater system	-

*R≠Zn: Roof without zinc; R=Zn: roof made with zinc; S: street; P: walking paths; Gs: garden above underground parking; G: garden

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Table 2. Characteristics of sampled rain events
(n is the number of rain events and h the cumulated water depth in mm)

Catchment	SS and organic matter		Organic micropollutants		Trace metals	
	n	h (mm)	n	h (mm)	n	h (mm)
Reference	8	57	9	81	5	36
North	7	65	5	58	9	99
Park	8	73	6	66	8	94
South	6	49	4	42	7	63

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Table 3. Runoff modelling parameters

Surface	IL (mm)	K_{inf} (mm/h)
Roof	0.33	
Street	0.65	0.2
Private garden	27.6	19.7

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2**Table 4.** Validation of micropollutant load simulations on the Reference site
(*t* for total fraction and *d* for dissolved fraction)

	Measured	Simulated		Measured	Simulated
SS (kg)	15.2 ± 1.5	15.4 ± 1.6	NP <i>t</i> (mg)	204 ± 18	202 ± 19
TOC (g)	2.08 ± 0.21	2.42 ± 0.28	NP <i>d</i> (mg)	142 ± 15	141 ± 13
DOC (g)	1.16 ± 0.12	1.38 ± 0.15	OP <i>t</i> (mg)	12.2 ± 1.3	11.7 ± 1.1
PCB <i>t</i> (µg)	994 ± 99	992 ± 79	OP <i>d</i> (mg)	8.51 ± 1.1	8.23 ± 0.83
PCB <i>d</i> (µg)	615 ± 92	695 ± 55	Cu (g)	4.50 ± 0.51	4.20 ± 0.41
PAH <i>t</i> (mg)	467 ± 42	434 ± 57	Zn (g)	146 ± 16	150 ± 15
PAH <i>d</i> (mg)	36.6 ± 2.9	34.9 ± 3.7	value ± uncertainty (80% confidence level)		

3

1 **Table 5.** Proportion of impervious surfaces and annual runoff coefficients
 2 (measured and simulated) (\pm measurement uncertainty)

	Reference	North	Park	South
Proportion of impervious surfaces	0.75 \pm 0.018	0.76 \pm 0.013	0.35 \pm 0.007	0.65 \pm 0.015
Measured annual runoff coefficient	0.72 \pm 0.086	0.40 \pm 0.048	0.15 \pm 0.018	0.32 \pm 0.038
Simulated annual runoff coefficient [*]	0.71 \pm 0.056	0.69 \pm 0.054	0.34 \pm 0.026	0.64 \pm 0.050
Annual runoff volume reduction attributed to SUDS		43%	54%	55%

* Simulated for a conventional stormwater pipe system as described in paragraph 2.4.1

3

Supplementary files:

Table S.1: log K_{ow} , analytical detection limits in both dissolved and particulate fraction

	Log K_{ow}	LQ in dissolved fraction (ng/L) for 2.5 L extracted	LQ in particulate fraction (ng/g.dw) for 100 mg extracted
Fluorene	4.14	0.8	20
Phenanthrene	4.46	0.4	10
Anthracene	4.54	0.8	20
Fluoranthene	5.22	1.2	30
Pyrene	5.18	1.4	35
Benzo[a]anthracene	5.91	1.6	15
Chrysene	5.61	0.6	10
Benzo[b]fluoranthene	5.80	0.4	10
Benzo[k]fluoranthene	6.00	0.4	10
Benzo[a]pyrene	6.04	0.4	10
Indeno[123]pyrene	7.00	0.6	15
Dibenzo[ah]anthracene	6.75	0.4	10
Benzo[ghi]perylene	7.23	0.4	10
PCB28	5.68	0.16	4
PCB52	5.68	0.42	10
PCB101	5.73	0.16	4
PCB118	6.41	0.16	4
PCB153	6.57	0.2	5
PCB138	6.79	0.16	4
PCB180	6.72	0.16	4
Para-tert-octylphenol	4.12	7.6	190
4-nonylphenol	4.48	3.2	330
MES	-	0.5 mg/L	
COD	-	0.5 mgC/L	-
COP	-	-	1.7 mgC/g

Table S.2: Analytical uncertainties at 80% confidence level

	Analytical uncertainties
MES	10 %
COD	5 %
COP	10 %
Σ HAP	29 %
Σ PCB	26 %
NP	23 %
OP	30 %
Cu	17 %
Pb	15 %
Zn	11 %

Annexe A.1: uncertainties calculation

Relative uncertainty (u_r) calculations of simulated or measured data were made according to the law of propagation of uncertainties.

In the following part: S is the surface, V is the volume, H is the rainfall, C is the concentration, CR is the runoff coefficient. The subscript i depends on the i^{th} event and the subscript j depends on the type of surface: p for pathways, s for street and b for building.

For measured rainfall H_i :

The uncertainty of measured rainfall is due to uncertainty of the pluviometer. V_{runnel} is the volume of the runnel which is equal to 0.2 mm and $u_r(V_{\text{runnel}})$ was estimated at 5.6 %. The relative uncertainty on the surface of the runnel, $u_r(S_{\text{runnel}})$, was estimated at 1 % by the constructor. N is the number of tipping of the runnel and $u_r(N_{\text{runnel}})$ was estimated at one over the whole rain.

$$u_r(H_i) = u_r\left(\frac{V_{\text{runnel}} \cdot N_{\text{runnel}}}{S_{\text{runnel}}}\right) = u_r(V_{\text{runnel}}) + u_r(S_{\text{runnel}}) + u_r(N_{\text{runnel}}) = 0.056 + 0.01 + \frac{0.2}{H_i}$$

For measured surface S :

The uncertainty of measured surface is due to uncertainty of the delimitation of the surface. The uncertainty on the delimitation of the boundaries was estimated to 0.5 m.

$$u_r(S) = u_r(l.l) = 2 \cdot u_r(l) = 2 \frac{0.5}{l} \approx 2 \frac{0.5}{\sqrt{S}} \approx \frac{1}{\sqrt{S}}$$

For simulated annual runoff coefficient (Table 5):

$$u_r^2(CR) = u_r^2(S) + \sum_i \frac{V_i^2}{V^2} [u_r^2(H_i) + u_r^2(CR_i)]$$

$$u_r^2(CR) \approx \frac{1}{S} + \sum_i \frac{V_i^2}{V^2} \left(\left(0.066 + \frac{0.2}{H_i} \right)^2 + u_r^2(CR_i) \right)$$

$u_r^2(CR_i)$ was assessed from the comparison between measured data and simulated data on Reference catchment. It was evaluated at 23 % if $H_i > 4$ mm and 47 % if $H_i < 4$ mm.

For simulated volumes (Figure 5):

$$u_r(V_j) = \sqrt{\frac{\sum_i V_{i,j} [u_r^2(H_i) + u_r^2(CR_{i,j})]}{V_j^2}}$$

$$\text{And } u_r(V) = \sqrt{\frac{u_r^2(V_p) \cdot V_p^2 + u_r^2(V_s) \cdot V_s^2 + u_r^2(V_b) \cdot V_b^2}{V}}$$

For simulated concentrations (Figure 5):

$$u_r(C_j) = \sqrt{\frac{\sum_i \left[u_r^2(C_{i,j}) \cdot \left(\frac{C_{i,j} \cdot V_{i,j}}{V_j} \right)^2 + (u_r^2(CR_{i,j}) + u_r^2(H_i) + u_r^2(S)) \cdot \frac{(V_{i,j} \cdot C_{i,j} \cdot (V_j - V_{i,j}))}{V_j^4} \right]}{C_j^2}}$$

$u_r^2(C_{i,j})$ is the relative uncertainty for the concentration of the event i and the type of surface j , evaluated according to Bertrand-Krajewski *et al.* (2001):

$$u_r^2(C_{i,j}) = u_r^2(C_{i,j}1) + u_r^2(C_{i,j}2) + u_r^2(C_{i,j}3) + u_r^2(C_{i,j}4) + u_r^2(C_{i,j}5) + u_r^2(C_{i,j}6)$$

with $u_r^2(C_{i,j}1)$ the uncertainty due to sampling $\approx 5\%$

with $u_r^2(C_{i,j}2)$ the uncertainty due to analysis given in Table S1 for each pollutant

with $u_r^2(C_{i,j}3)$ the uncertainty due to sub sampling $\approx 2\%$

with $u_r^2(C_{i,j}4)$ the uncertainty due to sampler installation $\approx 20\%$

with $u_r^2(C_{i,j}5)$ the uncertainty due to flow measurement $\approx 10\%$

with $u_r^2(C_{i,j}6)$ the uncertainty due to difference between the event duration and the sampling duration $\approx 20\%$

$$\text{And } u_r(C) = \frac{\sqrt{u_r^2(C_p) \cdot C_p^2 + u_r^2(C_s) \cdot C_s^2 + u_r^2(C_b) \cdot C_b^2}}{C}$$

For simulated masses (Figure 5):

$$u_r(M_j) = \sqrt{u_r^2(C_j) + u_r^2(V_j)}$$