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# 1 **Impact of urban pressure on the** 2 **spatial and temporal dynamics of** 3 **PAH fluxes in an urban tributary of** 4 **the Seine River (France)** 5

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20

## 21 Abstract

22 Polycyclic aromatic hydrocarbons (PAHs) produced by numerous anthropogenic activities are  
23 ubiquitous in the environment and have become a priority concern due to their potential  
24 severe biological impacts. A better understanding of PAH transfer at the catchment scale is

25 therefore necessary to improve the management of PAH contaminants and protect rivers.  
26 Furthermore, the impact of changes in hydrological regimes and land uses on PAH fluxes  
27 should be specifically investigated. Accordingly, the current research monitors the  
28 contamination in atmospheric fallout, soils and rivers in a 950-km<sup>2</sup> catchment (Orge River)  
29 characterized by an increasing urban gradient in downstream direction. During an entire  
30 hydrological year, river water contamination was quantified through regular sampling of both  
31 particulate and dissolved material at four river-monitoring stations, reflecting the increasing  
32 urbanization gradient. The significant input of PAHs from urban areas in downstream river  
33 sections corresponded to a specific PAH flux that reached 23 g km<sup>-2</sup> y<sup>-1</sup> despite the low  
34 sediment yield. Moreover, the comparison with runoff-specific fluxes reported in the literature  
35 underlined the major impact of urban runoff on the Orge River water and sediment quality.  
36 Nevertheless, the annual PAH load exported by the river (21 kg y<sup>-1</sup>) remained lower than the  
37 PAH inputs from atmospheric fallout (173 kg y<sup>-1</sup>), demonstrating the continuous  
38 accumulation of PAH from atmospheric fallout in the catchment soils. Consequently, the  
39 notably large PAH stock (close to 1000 tons) resulting from historical contamination of this  
40 early-industrialized region continues to increase due to ongoing atmospheric inputs.

41 Keywords: Urban runoff, PAH mass balance, atmospheric fallout, hydrological regime

42

## 43 1. Introduction

44 Urban sprawl and human activities have led to deleterious impacts on aquatic ecosystems,  
45 with systematic modification of the hydrology, biodiversity, and chemical quality of rivers  
46 referred to as “urban river syndrome” (Walsh et al., 2005). Organic contaminants such as  
47 polycyclic aromatic hydrocarbons (PAHs), which are released by incomplete combustion or  
48 pyrolysis of organic matter, have been targeted by environmental and health public agencies

49 since the 1970s due to their mutagenicity and carcinogenicity (Grimmer, 1985). Increasing  
50 concerns associated with the potential public health problems generated by widespread  
51 contamination with these substances has led to the implementation of policies and regulations  
52 to control the release of these hazardous compounds. In the European Union, the Water  
53 Framework Directives (WFD, Directive 2000/60/EC) requires all member states to adopt  
54 environmental measures for water bodies to reach a good ecological and chemical status. In  
55 France, the Seine River basin offers an emblematic example of contaminated areas, showing  
56 high PAH concentrations in rivers (up to 20 mg kg<sup>-1</sup> in river-transiting particles) (Fernandes et  
57 al., 1997; Uher et al., 2016) and complicating the achievement of good quality targets to such  
58 a large extent that the lowest quality grade in 2011 was attributed to the Seine River for those  
59 priority pollutants, including PAHs (Fisson et al., 2014).

60 Awareness of PAH impacts and their regulation occurred after two centuries of continuous  
61 release into the Western European atmosphere (Fernández et al., 2000), despite a decrease  
62 during the last several decades (Pacyna et al., 2003). This legacy deposition since the  
63 industrial revolution resulted in the accumulation of significant PAH stocks in the soils of the  
64 Seine River basin (Gaspéri et al., 2018; Gateuille et al., 2014a) as well as in other regions of  
65 Europe (Błońska et al., 2016; Jones et al., 1989). Soils are exposed to erosion processes  
66 (Lorgeoux et al., 2016; Rose and Rippey, 2002) that might supply PAH-contaminated  
67 particles to the rivers (Liu et al., 2013). Moreover, soils are currently exposed to additional  
68 PAH contamination sources originating from application of urban sewage sludge (Blanchard  
69 et al., 2007), direct vehicular exhaust gases (Gateuille et al., 2014b) and global atmospheric  
70 fallout associated with long-range transport of PAHs (Sofowote et al., 2011). The latter might  
71 be composed of PAHs originating from remote sources (Ding et al., 2007; Guzzella et al.,  
72 2016), local sources of vehicle exhaust emissions (Markiewicz et al., 2017; Oliveira et al.,  
73 2011), household heating (Motelay-Massei et al., 2004; Rocher et al., 2004) or industrial

74 inputs (Cetin et al., 2017; Motelay-Massei et al., 2007). Once deposited on the soil surface,  
75 PAHs can be transferred to the aquatic environments by soil erosion processes and runoff on  
76 impervious surfaces (Bomboi and Hernández, 1991; Krein and Schorer, 2000). In addition to  
77 atmospheric deposition, contaminated particles might be leached from urban surfaces (e.g.,  
78 from tires and roadway wears) before reaching water bodies (Van Metre et al., 2000). These  
79 complex transfer patterns among the atmosphere, soils and rivers are affected by land use and  
80 particularly by the occurrence of urban areas, which concentrate multiple PAH sources  
81 (Chalmers et al., 2007).

82 Numerous studies have improved our knowledge of PAH transfers in the environment  
83 although they separately addressed the atmosphere (Azimi et al., 2005), soils (Motelay-  
84 Massei et al., 2004), river water (Uher et al., 2016), and sediment (Liu et al., 2013). In  
85 contrast, few studies have investigated the global contamination at the catchment scale by  
86 integrating the atmosphere-soil-river continuum to assess the transfer of these contaminants  
87 and their persistence (Gateuille et al., 2014a; Rodenburg et al., 2010). Accordingly, few  
88 studies consider the spatial and temporal dynamics of PAH inputs and outputs at the  
89 catchment scale. Moreover, most of the available studies that estimate PAH loads supplied by  
90 urban areas only covered rather small surface areas (<1-300 ha) (Bressy et al., 2011;  
91 Hannouche et al., 2017). For larger catchments, identification of PAH inputs from urban areas  
92 was complicated by the remobilization of sediment stored in the channel (Schwientek et al.,  
93 2017). Therefore, additional research is needed to quantify both the supply and the signature  
94 of PAHs from urban areas to rivers.

95 The main objective of this study was to calculate the mass balance of PAH (inputs, exports  
96 and stocks) in a catchment affected by urban sprawl with a focus on the dynamics of riverine  
97 exports through sampling of river water and sediment during contrasting hydrological  
98 conditions. Based on this monitoring, the residence time of PAHs in the catchment was

99 estimated to supply new insights into the persistence of these contaminants in urban areas  
100 under temperate climate conditions.

101 To conduct the current research, a contaminated subcatchment of the Seine River basin, the  
102 Orge River (Froger et al., 2018; Le Pape et al., 2012), was selected based on its  
103 characteristics, which are similar to those of the Seine River, with a high urban pressure (up to  
104 5,000 inhabitants per km<sup>2</sup> for the Orge catchment at the outlet vs. approximately 8,000  
105 inhabitants per km<sup>2</sup> in the vicinity of Paris megacity). This catchment shows an increasing  
106 urbanization gradient in the downstream direction at the expense of cropland and forest.  
107 Wastewaters collected in the catchment are drained by a separate sewer to a treatment plant  
108 located in a nearby catchment, thus restricting the residential sources to run-off on urban  
109 areas, although limited discharges of untreated wastewater to the river might be caused by  
110 misconnections. These conditions offer an ideal site with which to investigate potential  
111 changes in PAH fluxes in response to urban sprawl and hydrological conditions.

112

## 113 2. Materials and methods

### 114 2.1. Study site

115 The study was conducted in the Orge River catchment, located 30 km south of Paris City (Fig.  
116 1; additional details can be found in Froger et al., (2018)). Three sampling sites were selected  
117 on the main stream of the Orge River, and a fourth site was located on its main tributary, the  
118 Yvette River. The proportion of urban areas strongly increases in the downstream direction,  
119 from 1% in the upper catchment to 56% at the outlet, and the proportion of agricultural lands  
120 decreases from 86% to 33% (Table S1). This change is reflected by the increase in population  
121 densities in the drainage areas of the monitoring stations, from 300 inhabitants per km<sup>2</sup> at the

122 most rural study site of Dourdan (“D”), to 5,000 inhabitants per km<sup>2</sup> at Viry (“V”, near the  
123 outlet).

124 The geology of the catchment is characterized by Eocene formations, including carbonate  
125 rocks, marls and gypsum, and Oligocene formations dominated by Fontainebleau sands (Le  
126 Pape et al., 2012).

127

## 128 2.2. Hydrological conditions

129 Five river water sampling campaigns were conducted from January to December 2016, those  
130 campaigns were organized to be as representative as possible of the various hydrological  
131 conditions observed in the river. During this period, water discharge measured at the outlet  
132 varied from 1.6 to 39.7 m<sup>3</sup> s<sup>-1</sup> with a mean of 3.5 m<sup>3</sup> s<sup>-1</sup> (Table S2). The sampling campaigns  
133 were organized to be representative of those hydrological conditions observed throughout the  
134 year in the Orge River (Fig. 2). Accordingly, each campaign was associated with a water flow  
135 class (Fig. 2). One campaign was organized during the low flow period (discharge < 2 m<sup>3</sup> s<sup>-1</sup>  
136 at the outlet), three campaigns during the mean water flow periods (2-5 m<sup>3</sup> s<sup>-1</sup>) and one  
137 campaign during high water flow (exceeding 5 m<sup>3</sup> s<sup>-1</sup>). Water samples were also collected on  
138 June 06, 2016 at every site (discharge of 15 m<sup>3</sup> s<sup>-1</sup>), during decrease of the water flow of June  
139 2016 exceptional flood extending from 05/29/2016 to 06/10/2016 (with water flow above 10  
140 m<sup>3</sup> s<sup>-1</sup>).

141

142

## 143 2.3. Sampling and PAH analysis

### 144 2.3.1. River and atmospheric deposition sampling

145 First, suspended particulate matter (SPM) was collected using sediment traps composed of  
146 metal bottles submerged in the river were deployed during 4 to 5 days to collect SPM by  
147 sedimentation for a longer period ( $n = 22$ ). Suspended particulate matter recovered from the  
148 traps was decanted, the supernatant was removed and the suspended solid was finally freeze-  
149 dried (Gateuille et al., 2014a). Samples of SPM collected using sediment traps were  
150 demonstrated as representative of suspended particles transported in the river, with PAH  
151 contamination levels similar to those of SPM collected by punctual sampling (Bartsch et al.,  
152 1996; Gateuille et al., 2014a).

153 In addition, punctual samples of river water were collected before and after each sediment  
154 traps installation ( $n = 32$ ), using 2-L brown glass bottles preliminarily washed with detergent  
155 (TFD4) and heated at 500°C. Extraction of dissolved PAH was rapidly conducted on the  
156 samples on the same day as the sample collection.

157 Suspended particulate matter concentrations were measured by filtering the two punctual bulk  
158 water samples collected during each campaign at each site, as well as two additional samples  
159 of 10 liters of river water, which provided four SPM estimations for each campaign. During  
160 the flood of June 2016 (Figure 2), punctual SPM samples ( $n = 4$ ) were collected on  
161 06/06/2016 to measure SPM content at each site.

162 Samples of total atmospheric fallout were collected near the Yvette monitoring site, in Orsay,  
163 from February 1, 2016 to March 30, 2017 (Fig. 1). Two funnels with 21 and 28 cm diameters  
164 were installed and connected to brown glass bottle collectors. Atmospheric fallout samples ( $n$   
165 = 26) were collected after variable periods (median: 16 days) depending on the amount of  
166 precipitation.



167

168

### 2.3.2. Sample preparation

169 The full analytical protocol (i.e, quantification method, and validation using certified  
170 material) used in previous studies (Bressy et al., 2012, 2011; Lorgeoux et al., 2016) is detailed  
171 in the supplementary material. In brief, instantaneous samples of river water and atmospheric  
172 fallout were filtered immediately after sampling, first on quartz filters GF/D (porosity of 2.7  
173  $\mu\text{m}$ ) and subsequently on GF/F (pore size of 0.47  $\mu\text{m}$ ) previously heated at 500°C. The filters  
174 were eventually freeze-dried. Following filtration, extraction was conducted using a solid-  
175 phase siliceous cartridge C18 (6 mL and 2000 mg). Solid sample extraction (atmospheric  
176 particles, SPM on filters and from sediment traps) was performed using microwave-assisted  
177 extraction (Microwave 3000, Anton Paar). Samples were introduced in closed bottles with a  
178 mixture of dichloromethane and methanol and heated at 100°C for 10 minutes (Lorgeoux et  
179 al., 2016). To avoid interference during the analysis, purification of all extracted samples was  
180 performed using silica gel columns (2.1 g of silica gel) to separate the different compound  
181 families using various solvent mixtures. Prior to extraction, an internal standard containing 6  
182 deuterated PAHs (naphthaleneD8, acenaphtheneD10, phenanthreneD10, pyreneD12,  
183 chryseneD12 and peryleneD12) was added to each sample, both in dissolved and particulate  
184 forms. The method was validated based on analyses of certified material (lake sediment  
185 standard NIST 1944).

186

187

### 2.3.3. PAH analysis

188 PAH concentrations in the dissolved and particulate phases of river water samples and  
189 atmospheric fallout were measured by gas chromatography coupled with a mass spectrometer  
190 (GC-MS, column RTX5SIL-MS (Restek, 60 m, 0.25 mm ID, 0.25  $\mu\text{m}$  df)) used in selected-

191 ion monitoring mode (see Suppl. Material for details). Helium was the vector gas, the  
192 injection volume was 1  $\mu\text{L}$ , and the samples were heated at 330°C for 6 minutes before  
193 analysis. The following PAH were measured: fluorene (Fl), phenanthrene (Phe), anthracene  
194 (Ant), fluoranthene (Flh), pyrene (Py), benzo(a)anthracene (BaA), chrysene (Chry),  
195 benzo(b)fluoranthene (BbF), benzo(k)fluoranthene (BkF), benzo(a)pyrene (BaP),  
196 indeno(1,2,3-cd)pyrene (IndP), dibenzo(a,h)anthracene (DbahA) and benzo(g,h,i)perylene  
197 (BghiP). The results were expressed as the sum of 13 PAHs. The results found for  
198 naphthalene, acenaphthylene and acenaphthene were not considered in the current research  
199 because these compounds are too volatile to be correctly quantified (Sicre et al., 2008).

## 200 2.4. Flux calculations

### 201 2.4.1. Daily riverine export

202 Daily fluxes of PAH exported by the Orge River were calculated based on Eqs. 1 and 2.

$$203 \quad \Phi_{part.,I,S} = [PAH]_{part.,I,S} \times [SPM]_{I,S} \times Q_{river,IS} \quad (1)$$

$$204 \quad \Phi_{diss.,I,S} = [PAH]_{diss.,I,S} \times Q_{river,IS} \quad (2)$$

205 where  $\Phi_{part.,I,S}$  and  $\Phi_{diss.,I,S}$  are the respective daily exports of the particulate and dissolved  
206 phases for campaign  $I$  in  $\text{g d}^{-1}$  at the site  $S$ ;  $[PAH]_{part.,I,S}$  is the concentration of PAH in  $\text{g t}^{-1}$  in  
207 the suspended particulate matter;  $[PAH]_{diss.,I,S}$  is the PAH concentration in the dissolved phase  
208 in  $\text{g m}^{-3}$ ;  $[SPM]_{I,S}$  is the total suspended matter concentration in  $\text{t m}^{-3}$  measured during  
209 campaign  $I$  for site  $S$ ; and  $Q_{river,IS}$  is the mean flow rate for site  $S$  during campaign  $I$  in  $\text{m}^{-3} \text{d}^{-1}$ .  
210 Fluxes were estimated using the median values of  $[PAH]_{part.,I,S}$ ,  $[SPM]_{I,S}$  and  $[PAH]_{diss.,I,S}$ .  
211 Accordingly, the mean values of  $\Phi_{part.,I,S}$  and  $\Phi_{diss.,I,S}$  and their respective standard deviations  
212 were quantified.

213 2.4.2. Annual fluxes

214 2.4.2.1. Atmospheric deposition

215 The PAH annual deposition is calculated using Eq. 3.

216 
$$\Phi_{an.dep} = \frac{\sum_{i=1}^n TAD}{\sum_{i=1}^n NbD \times S_{RG}} \times 365 \quad (3)$$

217 where  $\Phi_{an. dep}$  is the annual PAH deposition in  $\mu\text{g m}^{-2} \text{y}^{-1}$ ;  $\Sigma TAD$  is the sum of the total  
218 atmospheric deposition of the  $i^{th}$  sample in  $\mu\text{g}$ ;  $\Sigma NbD$  is the number of days between  
219 successive collections corresponding to the  $i^{th}$  sample; and  $S_{RG}$  is the surface area of the  
220 rainfall collector in  $\text{m}^2$ .

221

222 2.4.2.2. Riverine export

223 The mean annual flux exported with each fraction (dissolved and particulate) was calculated  
224 based on two types of equations: (1) based on data varying according to the hydrological  
225 regime and (2) using the SPM and PAH median concentrations. The combination of those  
226 estimations offers insight into the impact of the temporal variability of SPM and PAH  
227 measurements on the PAH final export.

228 The first set of equations (Eqs. 4 and 5) involves classification of the hydrological conditions  
229 (low, mean, high water flow periods and the extreme flood of June 2016), as described above  
230 (Section 2.2, Fig. 2).

231 
$$\Phi_{part,1,S} = \sum_{i=1}^n ([PAH]_{part,c,S} \times [SPM]_{c,S} \times Q_{i,c,S}) \quad (4)$$

232 
$$\Phi_{diss,1,S} = \sum_{i=1}^n ([PAH]_{diss,c,S} \times Q_{i,c,S}) \quad (5)$$

233 The annual PAH exports by the particulate phase  $\Phi_{part,1,S}$  and the dissolved phase  $\Phi_{diss,1,S}$  in g  
234  $\text{year}^{-1}$  at each site S were calculated as the sum of the fluxes estimated for each  $i$  day in  $\text{g d}^{-1}$ .

235 The daily flux was calculated using the median  $[PAH]_{part,c,S}$  in  $g\ t^{-1}$ ,  $[SPM]_{c,S}$  in  $t\ m^{-3}$  and  
 236  $[PAH]_{diss,c,S}$  in  $g\ m^{-3}$  from those campaigns corresponding to each water flow class  $c$  (defined  
 237 in Section 2.2) and daily water flow  $Q_{i,c,S}$  in  $m^3\ d^{-1}$  continuously measured near each sampling  
 238 site. To estimate PAH export during the flood of June 2016 (from 05/29/2016 to 06/10/2016),  
 239  $[SPM]$  measured at each site on 06/06/2016 and  $[PAH]_{part,c,S}$  corresponding to high water flow  
 240 periods were used.

241 The second set of equations (Eqs. 6 and 7) used to estimate the PAH riverine fluxes is based  
 242 on the median concentrations of PAH and SPM for each site:

$$243 \quad \Phi_{part,2,S} = \sum_{i=1}^n ([PAH]_{part,med,S} \times [SPM]_{med,S} \times Q_{i,S}) \quad (6)$$

$$244 \quad \Phi_{diss,2,S} = \sum_{i=1}^n ([PAH]_{diss,med,S} \times Q_{i,S}) \quad (7)$$

245 The annual PAH fluxes  $\Phi_{part,2,S}$  and  $\Phi_{diss,2,S}$  in  $g\ y^{-1}$  correspond to the sum of the  $i$  daily fluxes  
 246 at each site  $S$ . Those fluxes were calculated using the median PAH concentrations and  
 247 suspended particulate concentrations  $[PAH]_{part,med,S}$ ,  $[PAH]_{diss,med,S}$  and  $[SPM]_{med,S}$  expressed  
 248 respectively in  $\mu g\ g^{-1}$ ,  $ng\ L^{-1}$  and  $mg\ L^{-1}$ . For the exceptional flood of June 2016 (05/29/2016  
 249 – 06/10/2016),  $[SPM]$  measured the 06/06/2016 were used.

250 The mean PAH riverine export at each site  $S$  by the particulate phase ( $\Phi_{part,S}$ ) and the  
 251 associated standard deviation were calculated as the mean value of  $\Phi_{part,1,S}$  and  $\Phi_{part,2,S}$ . The  
 252 mean annual export by the dissolved phase ( $\Phi_{diss,S}$ ) was calculated from  $\Phi_{diss,1,S}$  and  $\Phi_{diss,2,S}$ .

#### 253 2.4.2.3. Specific particulate PAH flux

254 From the mean particulate PAH fluxes calculated using Eq. 4 and 6, specific fluxes were  
 255 estimated using the drainage area for each subcatchment and used in the following calculation  
 256 (Eq. 8).

$$257 \quad \Phi_{Spec,part,S} = \frac{(\Phi_{part,S})}{S_{DC,S}} \quad (8)$$

258 Therefore, at each site  $S$ , the specific particulate PAH flux in  $\text{g km}^{-2} \text{y}^{-1}$  was calculated using  
259 the annual particulate PAH export ( $\Phi_{\text{part},S}$ ) in  $\text{g y}^{-1}$  and the surface area drained at the site  $S$   
260  $S_{\text{DC},S}$ , in  $\text{km}^2$ . The calculations made for Viry site were considered to correspond to the total  
261 particulate PAH exported by the entire catchment.

262

#### 263 2.4.2.4. Specific sediment yield

264 The specific sediment yield was estimated for each sampling site based on Eq. 9:

$$265 \quad \Phi_{SY,S} = \frac{\sum_{i=1}^n ([SPM]_{\text{mean},S} \times Q_{i,S})}{SW} \quad (9)$$

266 The annual specific sediment yield  $\Phi_{SY,S}$  at site  $S$  in  $\text{t km}^{-2} \text{y}^{-1}$  was calculated as the sum of  
267 the daily SPM fluxes based on the mean concentration of SPM at site  $S$  in  $\text{mg L}^{-1}$   
268 ( $[SPM]_{\text{mean},S}$ ) based on the measurements conducted during the four campaigns (Figure 2) and  
269 the associated flow rate  $Q_{i,S}$  at day  $i$  across the surface of the watershed  $SW$  in  $\text{km}^2$ . For the  
270 exceptional flood of June 2016 (05/29/2016 – 06/10/2016),  $[SPM]$  measured on 06/06/2016  
271 were used.

272

#### 273 2.4.2.5. PAH stock in soils

274 The stock of PAH in soils under the three main land uses found in the study area (forests,  
275 cropland and urban areas) was calculated using Eq. 9 based on those PAH concentrations  
276 measured in soils (0-10 cm;  $n = 32$ ) collected across the entire Seine River basin (Gaspéri et  
277 al., 2018).

$$278 \quad \text{Stock} = [PAH] \times D_i \times \rho_{\text{soil}} \quad (9)$$

279 where Stock is the PAH stock in soils in  $\text{mg m}^{-2}$ ; [PAH] is the PAH concentration in  $\text{mg kg}^{-1}$   
280 ( $\sum 13\text{PAHs}$ ) of the soils;  $D_i$  is the depth fixed to 0.1 m; and  $\rho_{\text{soil}}$  is the bulk density of a typical  
281 soil of the region, i.e.,  $2000 \text{ kg m}^{-3}$  (Gaspéri et al., 2018).

282

### 283 3. Results and discussion

#### 284 3.1. PAH contents and fluxes

##### 285 3.1.1. Atmospheric fallout

286 Atmospheric fallout showed a wide range of variations in PAH ( $\sum 13\text{PAHs}$ ) contents, with  
287 particulate PAHs contents varying from 5 to  $93 \mu\text{g g}^{-1}$  and dissolved concentrations ranging  
288 from 30 to  $550 \text{ ng L}^{-1}$  (Table S6). The mean PAH concentration in the atmospheric deposition  
289 samples was  $161 \text{ ng L}^{-1}$  throughout the sampling period (min-max:  $89\text{-}2041 \text{ ng L}^{-1}$ ).  
290 Accordingly, the annual flux was estimated as  $182 \mu\text{g m}^{-2} \text{ y}^{-1}$  and was similar to the PAH  
291 atmospheric fallout measured at various locations in Europe (Table 1). Therefore, PAH  
292 atmospheric deposition measured at one single site in this study was considered to be  
293 representative of PAH atmospheric fallout across the entire Orge River catchment.

294

295 During the study period, no significant variation of PAH atmospheric deposition was  
296 observed in response to changes in temperature (Fig. S1 and S2 in SI), which contrasts with  
297 the results of previous studies showing higher PAH contents in atmospheric fallout collected  
298 during colder periods, which is generally attributed to household heating (Garban et al., 2002;  
299 Harrison et al., 1996; Motelay-Massei et al., 2003). Moreover, the rise in PAH production  
300 observed in winter is accentuated by a shift in the gas/particle distribution occurring at low  
301 temperatures and resulting in higher PAH contamination of particles (Kaupp and McLachlan,  
302 1999; J. Liu et al., 2013). The discrepancy between our results and those of previous studies

303 might be attributed to sampling duration differences (i.e., 2 weeks in this study and 2 to 7  
304 days in previous studies), which do not allow observation of finer temporal variations in PAH  
305 atmospheric fallout.

306 The proportion of PAH bound to particles varied from 16 to 95%, and the highest particulate  
307 proportion was observed in the sample with a total PAH concentration of 2041 ng L<sup>-1</sup>. The  
308 increase in particle-bound PAH from the first (February 2016 to July 2016) to the second  
309 period (July 2016 to March 2017) was probably due to the change in filtration material used  
310 in the current research (GF/F filters alone during period 1 vs. a combination of GF/D and  
311 GF/F filters during period 2) leading to a larger quantity of particles collected ( $9 \pm 5$  mg L<sup>-1</sup>  
312 for period 1 vs.  $22 \pm 26$  mg L<sup>-1</sup> for period 2). After this change, the particulate-bound PAH  
313 proportion systematically exceeded 50%. However, the PAH molecular composition of  
314 atmospheric particles remained stable with the dominance of 4-ring PAHs ( $51\% \pm 8\%$ )  
315 compared with that of 3- and 5-6-ring compounds, i.e.,  $22\% \pm 9\%$  and  $27\% \pm 10\%$  of the total  
316 PAH contamination, respectively. Phenanthrene (14%), fluoranthene (21%), pyrene (21%)  
317 and benzo(b)fluoranthene (10%) were the PAHs with the highest concentrations found in the  
318 particulate phase.

319

### 320 3.1.2. River contamination

321 The PAH concentrations in the dissolved phase ranged from 11 to 221 ng L<sup>-1</sup> with a median  
322 value of 41 ng L<sup>-1</sup> (Table 2). In the particulate phase, the PAH concentrations varied from 1.0  
323 to 45.2 µg g<sup>-1</sup> with a median of 4.6 µg g<sup>-1</sup>. Based on the suspended matter contents ranging  
324 from 2 to 68 mg L<sup>-1</sup>, depending on the site and the season considered, the estimated total river  
325 concentrations of particulate PAHs varied from 29 to 369 ng L<sup>-1</sup>.

326 Comparison with river contamination levels reported in the literature (Table 2) showed that  
327 the values found in the Orge River (mean:  $4.6 \mu\text{g g}^{-1}$ ) remained in the range of values  
328 measured in rivers impacted by urbanization (Seine:  $5.8 \mu\text{g g}^{-1}$ ; Elbe:  $4.7 \mu\text{g g}^{-1}$ ; Moselle:  $4.5$   
329  $\mu\text{g g}^{-1}$ ).

330

331 Particulate-bound PAHs supplied 57% of the total contamination (Fig. S3), in agreement with  
332 previous studies reporting that 3- to 6-ring PAHs are mainly transported with particles  
333 (Heemken et al., 2000; Shi et al., 2007). However, temporal variations in the PAH partition  
334 were observed, particularly at the Dourdan and Yvette sites. Furthermore, spatial variations  
335 were observed with an increase in particulate contribution in the downstream direction, with  
336 the highest particle-bound contribution of 72% found at the outlet at Viry station (Fig. 3b).  
337 These variations could not be related to variations in total suspended solid concentrations  
338 because SPM concentrations were similar between sites, integrating all campaigns (Dourdan:  
339  $14.1 \pm 15.5 \text{ mg L}^{-1}$ ; Egly:  $10.9 \pm 6.5 \text{ mg.L}^{-1}$ ; Yvette:  $15.5 \pm 9.6 \text{ mg.L}^{-1}$ ; Viry:  $10.2 \pm 15.8$   
340  $\text{mg.L}^{-1}$ ). Instead, this increase in particulate PAH proportions in the downstream direction was  
341 related to the spatial variations of particulate PAH concentrations increasing from  $3.3 \mu\text{g g}^{-1}$   
342 at the Dourdan site to  $8.5 \mu\text{g g}^{-1}$  at the Viry site (Fig. 3d). In addition, temporal variations  
343 associated with the change in hydrological conditions were observed (Fig. 3a), with the  
344 highest proportion of dissolved PAH observed in April and the lowest observed in January. A  
345 similar trend was observed for dissolved PAH concentrations (Fig. 3c) (Kruskal-Wallis test,  $\alpha$   
346  $<0.05$ ), suggesting that the change in the dissolved phase contamination might explain the  
347 temporal variations observed in PAH partitioning because SPM concentrations and  
348 particulate-bound PAH showed the absence of temporal variation (Fig. 3c). The higher  
349 dissolved PAH concentrations measured during high water flows might be explained by the  
350 increase in PAH desorption that occurs when particles are resuspended from the channel bed



351 and well mixed in the water column during floods (Belles et al., 2016). However, although  
352 this increase in surface water contamination in response to flood events has been observed in  
353 several studies (Dong et al., 2016; Mouhri et al., 2008), it has not been systematically reported  
354 (Gateuille et al., 2014a; Sicre et al., 2008).

355

### 356 3.2. Impact of hydrological conditions on PAH fluxes

357 The daily riverine exports of dissolved PAHs were estimated based on only four sampling  
358 campaigns for the dissolved PAHs (Fig. 2). Detailed values are supplied in the supplementary  
359 material. The fluxes of dissolved PAH at the outlet ranged from 6 to 32 g d<sup>-1</sup>, with the highest  
360 load observed during the flood of April 2016 and the lowest load observed during the low  
361 stage period of August 2016. Compared with those results obtained for the Seine River, which  
362 are characterized by a discharge varying from 100 to 700 m<sup>3</sup> s<sup>-1</sup> and dissolved PAH fluxes  
363 estimated to vary from 690 to 4838 g d<sup>-1</sup> (PAH concentration of 80 ng L<sup>-1</sup>), the Orge River  
364 supplied 1% of the total dissolved PAH export of the Seine River basin, which remains  
365 proportional to its surface (1.3% of the Seine River basin).

366 Fluxes from each upper subcatchment (i.e., Egly and Yvette) are shown in Fig. 4.  
367 Accordingly, the difference between the fluxes calculated for the entire catchment and the  
368 upper subcatchment fluxes corresponded to those PAH fluxes supplied by the downstream  
369 urban area that is referred to as Viry<sub>sub</sub> in the remainder of the text. With respect to the  
370 subcatchment fluxes, it appears that the main proportion of the dissolved PAH exported at the  
371 outlet (Viry) originated from upper catchment portions (i.e., Egly and Yvette subcatchments)  
372 because their PAH fluxes were equivalent to those fluxes measured at the outlet, resulting in a  
373 budget of +6.0 g d<sup>-1</sup>, -12.6 g d<sup>-1</sup>, -0.5 g d<sup>-1</sup> and -0.2 g d<sup>-1</sup> for January, April, August and  
374 November 2016, respectively. This result demonstrates the absence of significant dissolved

375 PAH supply from the Viry<sub>sub</sub> subcatchment suggesting that PAH inputs from those urban  
376 areas downstream were mainly associated with the particulate phase.

377 In contrast, the temporal variability of particulate PAH fluxes suggests the large impact of  
378 seasonal variations of water flow and urbanization on PAH discharge. Accordingly, the  
379 highest PAH loads were observed during the flood in April, corroborating the results of  
380 previous studies showing a major export of PAH during floods (Conaway et al., 2013; Mouhri  
381 et al., 2008). During this flood event, particulate PAH supplied to the lower river section that  
382 drains urban areas, i.e., Viry<sub>sub</sub>, reached a maximum of 23.9 g d<sup>-1</sup>. This result suggests that the  
383 downstream portion of the catchment concentrating the majority of the urban areas (Fig. 1)  
384 supplied up to 53% of the total PAH exported by the particulate phase in the catchment for  
385 this campaign (Table 3). During the moderate water flow periods of January and November  
386 2016, the Viry<sub>sub</sub> loads were 18.9 and 17.0 g d<sup>-1</sup>, respectively, supplying 62% of the total PAH  
387 flux at the outlet (Table 3). A negative export from this urban river section was observed  
388 during the low flow period of August 2016, with -4.0 g d<sup>-1</sup> underlining the notably low supply  
389 of PAH from these areas during this campaign.

390 Comparison of the different subcatchment (i.e., Egly, Yvette and Viry<sub>sub</sub>) contributions to the  
391 total particulate PAH fluxes clearly shows that Viry<sub>sub</sub> supplied a significant contribution of  
392 PAHs in January, April and November despite the low total sediment load (Table 3). Studies  
393 conducted in urban areas reported that PAH contamination in runoff waters was mainly  
394 carried by the particulate phase (85%) and that it did not decrease with increasing stormflow,  
395 which suggests that urban surfaces supply a constant source of particulate PAH (Gasperi et  
396 al., 2009; Hwang and Foster, 2006). These observations are corroborated by the results found  
397 in the current research showing that the occurrence of heavy rainfall generated higher runoff  
398 volumes in urban areas (e.g., during the flood event in April 2016 and the average discharge

399 conditions campaigns of January and November 2016), thus increasing the PAH load supplied  
400 by these areas (Table 3).

401 In contrast to Viry<sub>sub</sub>, the higher sediment yield of the Yvette subcatchment was associated  
402 with the lowest contribution to the PAH total export at the outlet (Table 3). This result is  
403 likely explained by the construction of an extensive research campus, the Saclay Plateau, in  
404 the upper catchment portions, resulting in extensive soil erosion (Chin, 2006; Huon et al.,  
405 2017) and the dilution of PAH concentrations by preferential mobilization of uncontaminated  
406 subsurface particles. Among the four campaigns, only the survey conducted during the low  
407 flow period in August 2016 showed the major contribution of the upper catchment portions to  
408 the total PAH export. The negative mass-balance for this campaign could be explained by  
409 both sedimentation of particles during their transfer from Egly and Yvette to the outlet and a  
410 notably low input from Viry<sub>sub</sub>, insufficient to compensate the deposition of upstream PAH.  
411 Finally, the impact of urban runoff on PAH fluxes is particularly clear in the downstream  
412 portion of the Orge River catchment because of the channelization of the river section near the  
413 outlet. This configuration limits sediment storage and potential dilution of PAH urban loads  
414 by particles eroded from less contaminated soils or channel banks (Schwientek et al., 2017,  
415 2013).

416

417

### 418 3.3. Annual mass balance of PAH

419 The annual PAH catchment budget was estimated for 2016 based on PAH concentrations,  
420 SPM concentrations, daily water flow and total atmospheric deposition (Fig. 5). First, global  
421 accumulation of these compounds was observed. From the 173 kg of PAHs deposited with  
422 atmospheric fallout on catchment soils, only  $34.5 \pm 0.7$  kg of PAH were exported by the river,

423 resulting in an accumulation of 80% of the contamination deposited from the atmosphere in  
424 the soils ( $138 \text{ kg y}^{-1}$ ). A similar result was found for all subcatchments, with an accumulation  
425 of  $28 \text{ kg y}^{-1}$  for the Yvette catchment,  $6 \text{ kg y}^{-1}$  for the Dourdan catchment,  $73 \text{ kg y}^{-1}$  for the  
426 Egly catchment and  $31 \text{ kg y}^{-1}$  for Viry<sub>sub</sub>, corresponding to 84%, 87%, 84% and 68% of the  
427 respective atmospheric fallout inputs. The PAH accumulation rates are based on the  
428 hypothesis of conservation of PAH deposited by atmospheric fallout in soils. This assumption  
429 is supported by the higher PAH stability of those 4-to-6 ring PAHs (Biache et al., 2014;  
430 Cébron et al., 2013), contributing to 66% of the total atmospheric fallout. A second  
431 assumption is that atmospheric deposition is homogenous across the catchment. This later  
432 hypothesis is supported by the similar PAH deposition rates measured across Europe and  
433 particularly across the entire Paris region (Table 1). The accumulation rates estimated in the  
434 current study should be considered as an order of magnitude given the large uncertainties  
435 associated with the particle flux estimations and the assumptions made on the PAH behavior.

436 In both atmospheric fallout and riverine exports, the particulate fraction dominated the  
437 transport of PAH, especially in the Yvette subcatchment (82% of particulate-bound PAH) and  
438 in the lower portions of the catchment (Viry<sub>sub</sub>: 93%). Specific particulate PAH fluxes were  
439 calculated, demonstrating that Viry<sub>sub</sub> was the main source of particulate PAH, with a  
440 production of  $46.7 \pm 2.5 \text{ g km}^{-2} \text{ y}^{-1}$  (Table 4) compared with the total PAH export at the outlet  
441 in Viry ( $28.7 \pm 0.1 \text{ g km}^{-2} \text{ y}^{-1}$ ) and the Egly and Yvette subcatchments ( $19.1 \pm 1.0$  and  $24.8 \pm$   
442  $0.7 \text{ g km}^{-2} \text{ y}^{-1}$  respectively). Specific PAH fluxes and sediments yields were estimated for a  
443 compilation of studies found in the literature (Table 2; Table 4). Annual specific loads found  
444 in the current research ( $28.7 \text{ g km}^{-2} \text{ y}^{-1}$ ) were higher than those fluxes reported for the Moselle  
445 River ( $13.1 \text{ g km}^{-2} \text{ y}^{-1}$ ), and appeared to be similar to the loads reported for the Orgeval River  
446 ( $30 \text{ g km}^{-2} \text{ y}^{-1}$ ), the Elbe River ( $28.5 \text{ g km}^{-2} \text{ y}^{-1}$ ) and the Mackenzie River ( $27 \text{ g km}^{-2} \text{ y}^{-1}$ ).  
447 However, among those sites with equivalent PAH specific loads, the rural subcatchments of

448 the Orgeval and Mackenzie Rivers showed a sediment yield 3-fold higher than the other  
449 catchments. Despite their lower sediment yields, the urban Orge, Moselle and Elbe River  
450 catchments were characterized by relatively high PAH riverine export, reflecting the high  
451 contamination levels found in the sediment transiting these low-turbidity rivers. Accordingly,  
452 the impact of urban inputs on the contamination of the river can be characterized by a ratio  
453 between the PAH specific load (in  $\text{g km}^{-2} \text{y}^{-1}$ ) and the sediment yield (in  $\text{t km}^{-2} \text{y}^{-1}$ ), resulting  
454 in a theoretical PAH concentration in SPM (in  $\text{g.t}^{-1}$ ), considering both the anthropogenic  
455 pressure and the potential dilution in the river (Table 4). This ratio is similar to the mean  
456 particle-bound PAH concentration developed by Schwientek et al., (2017).

457 Consequently, urban catchments, regardless of their drainage surface (i.e., Orge River, Seine  
458 River, Moselle River, Körsh River and Elbe River), showed ratios exceeding 4, whereas rural  
459 catchments such as the Orgeval River and forested catchments such as the Mackenzie River  
460 were characterized by ratios lower than 2. The Rhône River, a mixed catchment that drains  
461 dense urban areas (Lyon) as well as large forested and agricultural areas in the upper portions,  
462 showed a low ratio of 2.2, explained by the notably high sediment yield ( $76 \text{ t km}^{-2} \text{y}^{-1}$ ). In the  
463 Orge River catchment, this ratio increased with the urbanization gradient in the downstream  
464 direction (Dourdan < Yvette < Egly < Viry<sub>sub</sub>), confirming the higher supply of PAHs from  
465 urban areas.

466 The ratio of 15 found for Viry<sub>sub</sub> was similar to that found for precipitation runoff collected in  
467 the small urban catchment of Sucy, located south of Paris (Hannouche et al., 2017). This  
468 catchment showed characteristics similar to those found in Viry<sub>sub</sub> (population density of 2500  
469 inhabitants per  $\text{km}^2$  and dense urban area) with a load ratio of 17 (Table 4). Therefore, the  
470 similarities between PAH inputs from Viry<sub>sub</sub> and those PAH loads from runoff on urban  
471 surfaces illustrate the impact of urban runoff on the Orge River quality. These results also  
472 suggest that this ratio reflects the impact of both historical and current human activities

473 supplying contaminants to the river on the sediment quality and might offer a tool for  
474 environmental assessment of the anthropogenic pressure on rivers. Based on the data collected  
475 in the current research and those reported in the literature (Table 4), a maximum ratio value of  
476 2 is proposed as a threshold to characterize catchments with a low impact of human activities  
477 on the river quality. Furthermore, ratio values exceeding 4 are suggested for identification of  
478 highly impacted catchments. Nevertheless, to better constrain PAH riverine export,  
479 continuous measurements of SPM should be conducted to refine PAH fluxes estimations and  
480 specific sediment yields in the catchment.

481

482

#### 483 3.4. Evaluation of PAH stocks in soils depending on land use

484 Stocks of PAHs in the upper 10-cm layer of the soils were estimated under each land use  
485 (Table S8) based on the values measured in a set of soils collected across the Paris areas  
486 including the Orge River catchment area (Gaspéri et al., 2018). However, as a large range of  
487 PAH concentrations was reported in Gasperi et al. (2018), extreme values (min and max)  
488 were also considered when estimating the PAH stock in the soils of the Orge River catchment  
489 to evaluate the range of variability of the stock estimate. The application of these values to the  
490 soils of the Orge River catchment resulted in a significant stock of PAH in soils (Table 5),  
491 with a minimum of 7, 13 and 14 tons of PAHs in urban soils, 13 tons in forest soils, and 14  
492 tons in agricultural soils. Atmospheric fallout of 46, 53 and 74 kg y<sup>-1</sup> deposited on urban,  
493 forested and agricultural soils, appeared to be negligible (<0.01%) compared to the estimated  
494 PAH stock in the soils corresponding to each of the three land uses considered (Table 8).  
495 These levels of soil contamination are highly likely to be associated with the accumulation of  
496 PAH emitted for more than a century by industrial and urban activities in the region

497 (Lorgeoux et al., 2016; Pacyna et al., 2003). If assuming that the current PAH deposition  
498 measured in the current research (i.e.,  $182 \mu\text{g m}^{-2} \text{y}^{-1}$ ) is representative of that observed over  
499 the long term, 161 years is expected to accumulate the lowest PAH stock estimated in soils,  
500 whereas 61,000 years are required to reach those values found in the most polluted soils.  
501 Those estimations suggest that higher PAH atmospheric deposition rates occurred during the  
502 late 19<sup>th</sup> and the early 20<sup>th</sup> centuries, which corroborates the results of a study conducted in  
503 Northern Spain that reported atmospheric deposition rates of up to  $2,500 \mu\text{g m}^{-2} \text{y}^{-1}$  in the  
504 1970s (Leorri et al., 2014).

505 Despite the decrease of long-range atmospheric PAH fallout (Pacyna et al., 2003), the  
506 reservoir of legacy PAH contamination in soils is not declining, as demonstrated by the  
507 ongoing PAH accumulation observed in 2016 in soils of the Orge River catchment.  
508 Significant PAH stocks have also been observed in the soils in rural areas of the Seine River  
509 basin, confirming the long term accumulation of PAH in the entire region (Gateuille et al.,  
510 2014a). Despite multiple studies demonstrating PAH sensitivity to biodegradation or  
511 oxidation processes in soils and rivers (Biache et al., 2014; Haritash and Kaushik, 2009), the  
512 PAH stocks found in soils suggest low degradation of PAH in natural environments.  
513 Consequently, management of the current PAH contamination in ecosystems should consider  
514 the dominance of this legacy contamination and focus on those PAH transfers and pathways  
515 to the aquatic system in addition to mitigation of local sources such as vehicular exhaust and  
516 household heating.

517

## 518 4. Conclusions

519 The current study quantified PAH contamination in the atmospheric fallout during an entire  
520 year, and the contamination of main rivers of the Orge River catchment based on five

521 campaigns representative of those hydrological variations observed in this area. Moreover, the  
522 temporal variations of the PAH fluxes and those of particulate PAH concentrations  
523 demonstrated the significant contribution of urban areas in the lower catchment portions to the  
524 particulate PAH export, which primarily occurred during mean and high water flow periods.  
525 The ratio of PAH specific fluxes and sediment yields, which was developed in the current  
526 research as an indicator of anthropogenic pressure on catchments, confirmed the impact of  
527 urbanization on the Orge River. However, despite the large PAH exports from urban areas,  
528 PAH continues to accumulate in the catchment soils. Finally, comparison of the current PAH  
529 deposition with the estimation of the PAH stock in the catchment soils based on values from  
530 the literature suggests that legacy contamination of the soils might act as an extensive source  
531 of PAH supplied to the rivers during the next several decades or even centuries. Source  
532 apportionment should therefore be conducted to quantify the respective contributions of  
533 legacy and contemporary contamination to PAH transiting the rivers. The identification of  
534 PAH sources using multiple tracing tools, including PAH diagnostic ratios, might offer  
535 further insights into the contamination dynamics in early industrialized catchments and guide  
536 the design of effective measures to manage this ubiquitous contamination.

537

538

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# References

- Azimi, S., Rocher, V., Muller, M., Moilleron, R., Thevenot, D.R., 2005. Sources, distribution and variability of hydrocarbons and metals in atmospheric deposition in an urban area (Paris, France). *Sci. Total Environ.* 337, 223–239.  
<https://doi.org/10.1016/j.scitotenv.2004.06.020>
- Bartsch, L. a., Rada, R.G., Sullivan, J.F., 1996. A comparison of solids collected in sediment traps and automated water samplers. *Hydrobiologia* 323, 61–66.  
<https://doi.org/10.1007/BF00020547>
- Belles, A., Mamindy-Pajany, Y., Alary, C., 2016. Simulation of aromatic polycyclic hydrocarbons remobilization from a river sediment using laboratory experiments supported by passive sampling techniques. *Environ. Sci. Pollut. Res.* 23, 2426–2436.  
<https://doi.org/10.1007/s11356-015-5462-y>
- Biache, C., Mansuy-Huault, L., Faure, P., 2014. Impact of oxidation and biodegradation on the most commonly used polycyclic aromatic hydrocarbon (PAH) diagnostic ratios: Implications for the source identifications. *J. Hazard. Mater.* 267, 31–39.  
<https://doi.org/10.1016/j.jhazmat.2013.12.036>
- Blanchard, M., Teil, M.J., Guigon, E., Larcher-Tiphagne, K., Ollivon, D., Garban, B., Chevreuil, M., 2007. Persistent toxic substance inputs to the river Seine basin (France) via atmospheric deposition and urban sludge application. *Sci. Total Environ.* 375, 232–243. <https://doi.org/10.1016/j.scitotenv.2006.12.012>
- Błońska, E., Lasota, J., Szuszkiewicz, M., Łukasik, A., Klamerus-Iwan, A., 2016. Assessment of forest soil contamination in Krakow surroundings in relation to the type of stand.

- Environ. Earth Sci. 75, 1205. <https://doi.org/10.1007/s12665-016-6005-7>
- Bomboi, M.T., Hernández, A., 1991. Hydrocarbons in urban runoff: Their contribution to the wastewaters. *Water Res.* 25, 557–565. [https://doi.org/10.1016/0043-1354\(91\)90128-D](https://doi.org/10.1016/0043-1354(91)90128-D)
- Bressy, A., Gromaire, M.-C., Lorgeoux, C., Chebbo, G., 2011. Alkylphenols in atmospheric depositions and urban runoff. *Water Sci. Technol.* 63, 671 LP-679.
- Bressy, A., Gromaire, M.-C., Lorgeoux, C., Saad, M., Leroy, F., Chebbo, G., 2012. Towards the determination of an optimal scale for stormwater quality management: Micropollutants in a small residential catchment. *Water Res.* 46, 6799–6810. <https://doi.org/10.1016/j.watres.2011.12.017>
- Cébron, A., Faure, P., Lorgeoux, C., Ouvrard, S., Leyval, C., 2013. Experimental increase in availability of a PAH complex organic contamination from an aged contaminated soil: Consequences on biodegradation. *Environ. Pollut.* 177, 98–105. <https://doi.org/10.1016/j.envpol.2013.01.043>
- Cetin, B., Ozturk, F., Keles, M., Yurdakul, S., 2017. PAHs and PCBs in an Eastern Mediterranean megacity, Istanbul: Their spatial and temporal distributions, air-soil exchange and toxicological effects. *Environ. Pollut.* 220, 1322–1332. <https://doi.org/10.1016/j.envpol.2016.11.002>
- Chalmers, A.T., Van Metre, P.C., Callender, E., 2007. The chemical response of particle-associated contaminants in aquatic sediments to urbanization in New England, U.S.A. *J. Contam. Hydrol.* 91, 4–25. <https://doi.org/10.1016/j.jconhyd.2006.08.007>
- Chin, A., 2006. Urban transformation of river landscapes in a global context. *Geomorphology* 79, 460–487. <https://doi.org/10.1016/j.geomorph.2006.06.033>

- Conaway, C.H., Draut, A.E., Echols, K.R., Storlazzi, C.D., Ritchie, A., 2013. Episodic Suspended Sediment Transport and Elevated Polycyclic Aromatic Hydrocarbon Concentrations in a Small, Mountainous River in Coastal California. *River Res. Appl.* 29, 919–932. <https://doi.org/10.1002/rra.2582>
- Delmas, M., Pak, L.T., Cerdan, O., Souchère, V., Le Bissonnais, Y., Couturier, A., Sorel, L., 2012. Erosion and sediment budget across scale: A case study in a catchment of the European loess belt. *J. Hydrol.* 420–421, 255–263. <https://doi.org/10.1016/j.jhydrol.2011.12.008>
- Ding, X., Wang, X.-M., Xie, Z.-Q., Xiang, C.-H., Mai, B.-X., Sun, L.-G., Zheng, M., Sheng, G.-Y., Fu, J.-M., Pöschl, U., 2007. Atmospheric polycyclic aromatic hydrocarbons observed over the North Pacific Ocean and the Arctic area: Spatial distribution and source identification. *Atmos. Environ.* 41, 2061–2072. <https://doi.org/10.1016/j.atmosenv.2006.11.002>
- Dong, J., Xia, X., Wang, M., Xie, H., Wen, J., Bao, Y., 2016. Effect of recurrent sediment resuspension-deposition events on bioavailability of polycyclic aromatic hydrocarbons in aquatic environments. *J. Hydrol.* 540, 934–946. <https://doi.org/10.1016/j.jhydrol.2016.07.009>
- Fernandes, M.B., Sicre, M.-A., Boireau, A., Tronczynski, J., 1997. Polyaromatic hydrocarbon (PAH) distributions in the Seine River and its estuary. *Mar. Pollut. Bull.* 34, 857–867. [https://doi.org/10.1016/S0025-326X\(97\)00063-5](https://doi.org/10.1016/S0025-326X(97)00063-5)
- Fernández, P., Vilanova, R.M., Martínez, C., Appleby, P., Grimalt, J.O., 2000. The Historical Record of Atmospheric Pyrolytic Pollution over Europe Registered in the Sedimentary PAH from Remote Mountain Lakes. *Environ. Sci. Technol.* 34, 1906–1913.

<https://doi.org/10.1021/es9912271>

Fisson, C., Bacq, N., Gilles, B., Boust, D., Budzinski, H., Jean, D., Fontaine, T., Garnier, J., Homer, S., Lacroix, J., Leroy, B., Sansom, S., 2014. Qualité des eaux de l'Estuaire de la Seine - Fascicule 3.2.

Foan, L., Domercq, M., Bermejo, R., Santamaría, J.M., Simon, V., 2015. Mosses as an integrating tool for monitoring PAH atmospheric deposition: Comparison with total deposition and evaluation of bioconcentration factors. A year-long case-study. *Chemosphere* 119, 452–458. <https://doi.org/10.1016/j.chemosphere.2014.06.071>

Froger, C., Ayrault, S., Evrard, O., Monvoisin, G., Bordier, L., Lefèvre, I., Quantin, C., 2018. Tracing the sources of suspended sediment and particle-bound trace metal elements in an urban catchment coupling elemental and isotopic geochemistry, and fallout radionuclides. *Environ. Sci. Pollut. Res.* 25, 28667–28681. <https://doi.org/10.1007/s11356-018-2892-3>

Garban, B., Blanchoud, H., Motelay-Massei, A., Chevreuil, M., Ollivon, D., 2002. Atmospheric bulk deposition of PAHs onto France: trends from urban to remote sites. *Atmos. Environ.* 36, 5395–5403. [https://doi.org/10.1016/S1352-2310\(02\)00414-4](https://doi.org/10.1016/S1352-2310(02)00414-4)

Gaspéri, J., Ayrault, S., Moreau-Guigon, E., Alliot, F., Labadie, P., Budzinski, H., Blanchard, M., Muresan, B., Caupos, E., Cladière, M., Gateuille, D., Tassin, B., Bordier, L., Teil, M.-J., Bourges, C., Desportes, A., Chevreuil, M., Moilleron, R., 2018. Contamination of soils by metals and organic micropollutants: case study of the Parisian conurbation. *Environ. Sci. Pollut. Res.* 25, 23559–23573. <https://doi.org/10.1007/s11356-016-8005-2>

Gasper, J., Garnaud, S., Rocher, V., Moilleron, R., 2009. Priority pollutants in surface waters

- and settleable particles within a densely urbanised area: Case study of Paris (France). *Sci. Total Environ.* 407, 2900–2908. <https://doi.org/10.1016/j.scitotenv.2009.01.024>
- Gateuille, D., Evrard, O., Lefevre, I., Moreau-Guigon, E., Alliot, F., Chevreuil, M., Mouchel, J.-M., 2014a. Mass balance and decontamination times of Polycyclic Aromatic Hydrocarbons in rural nested catchments of an early industrialized region (Seine River basin, France). *Sci. Total Environ.* 470–471, 608–617. <https://doi.org/10.1016/j.scitotenv.2013.10.009>
- Gateuille, D., Evrard, O., Lefevre, I., Moreau-Guigon, E., Alliot, F., Chevreuil, M., Mouchel, J.M., 2014b. Combining measurements and modelling to quantify the contribution of atmospheric fallout, local industry and road traffic to PAH stocks in contrasting catchments. *Environ. Pollut.* 189, 152–160. <https://doi.org/10.1016/j.envpol.2014.02.029>
- Gocht, T., Klemm, O., Grathwohl, P., 2007. Long-term atmospheric bulk deposition of polycyclic aromatic hydrocarbons (PAHs) in rural areas of Southern Germany. *Atmos. Environ.* 41, 1315–1327. <https://doi.org/10.1016/j.atmosenv.2006.09.036>
- Grimmer, G., 1985. PAH—Their contribution to the carcinogenicity of various emissions†. *Toxicol. Environ. Chem.* 10, 171–181. <https://doi.org/10.1080/02772248509357101>
- Guzzella, L., Salerno, F., Freppaz, M., Roscioli, C., Pisanello, F., Poma, G., 2016. POP and PAH contamination in the southern slopes of Mt. Everest (Himalaya, Nepal): Long-range atmospheric transport, glacier shrinkage, or local impact of tourism? *Sci. Total Environ.* 544, 382–390. <https://doi.org/10.1016/j.scitotenv.2015.11.118>
- Hannouche, A., Chebbo, G., Joannis, C., Gasperi, J., Gromaire, M., Moilleron, R., Barraud, S., Ruban, V., 2017. Stochastic evaluation of annual micropollutant loads and their

uncertainties in separate storm sewer. *Environ. Sci. Pollut. Res.* 1–15.

Haritash, A.K., Kaushik, C.P., 2009. Biodegradation aspects of Polycyclic Aromatic Hydrocarbons (PAHs): A review. *J. Hazard. Mater.* 169, 1–15. <https://doi.org/10.1016/j.jhazmat.2009.03.137>

Harrison, R.M., Smith, D.J.T., Luhana, L., 1996. Source apportionment of atmospheric polycyclic aromatic hydrocarbons collected from an urban location in Birmingham, U.K. *Environ. Sci. Technol.* 30, 825–832 ST–Source apportionment of atmospheric.

Heemken, O.P., Stachel, B., Theobald, N., Wenclawiak, B.W., 2000. Temporal variability of organic micropollutants in suspended particulate matter of the River Elbe at Hamburg and the River Mulde at Dessau, Germany. *Arch. Environ. Contam. Toxicol.* 38, 11–31. <https://doi.org/10.1007/s002449910003>

Huon, S., Evrard, O., Gourdin, E., Lefèvre, I., Bariac, T., Reyss, J.-L., Henry des Tureaux, T., Sengtaheuanghong, O., Ayrault, S., Ribolzi, O., 2017. Suspended sediment source and propagation during monsoon events across nested sub-catchments with contrasted land uses in Laos. *J. Hydrol. Reg. Stud.* 9, 69–84. <https://doi.org/10.1016/j.ejrh.2016.11.018>

Hwang, H.-M., Foster, G.D., 2006. Characterization of polycyclic aromatic hydrocarbons in urban stormwater runoff flowing into the tidal Anacostia River, Washington, DC, USA. *Environ. Pollut.* 140, 416–426. <https://doi.org/10.1016/j.envpol.2005.08.003>

Jones, K.C., Stratford, J.A., Tidridge, P., Waterhouse, K.S., Johnston, A.E., 1989. Polynuclear aromatic hydrocarbons in an agricultural soil: Long-term changes in profile distribution. *Environ. Pollut.* 56, 337–351. [https://doi.org/10.1016/0269-7491\(89\)90079-1](https://doi.org/10.1016/0269-7491(89)90079-1)

Kaupp, H., McLachlan, M.S., 1999. Gas/particle partitioning of PCDD/Fs, PCBs, PCNs and

PAHs. *Chemosphere* 38, 3411–3421. [https://doi.org/10.1016/S0045-6535\(98\)00554-2](https://doi.org/10.1016/S0045-6535(98)00554-2)

Krein, A., Schorer, M., 2000. Road runoff pollution by polycyclic aromatic hydrocarbons and its contribution to river sediments. *Water Res.* 34, 4110–4115. [https://doi.org/10.1016/S0043-1354\(00\)00156-1](https://doi.org/10.1016/S0043-1354(00)00156-1)

Le Meur, M., Mansuy-Huault, L., Lorgeoux, C., Bauer, A., Gley, R., Vantelon, D., Montargès-Pelletier, E., 2017. Spatial and temporal variations of particulate organic matter from Moselle River and tributaries: A multimolecular investigation. *Org. Geochem.* 110, 45–56. <https://doi.org/10.1016/j.orggeochem.2017.04.003>

Le Pape, P., Ayrault, S., Quantin, C., 2012. Trace element behavior and partition versus urbanization gradient in an urban river (Orge River, France). *J. Hydrol.* 472–473, 99–110. <https://doi.org/10.1016/j.jhydrol.2012.09.042>

Leorri, E., Mitra, S., Irabien, M.J., Zimmerman, A.R., Blake, W.H., Cearreta, A., 2014. A 700year record of combustion-derived pollution in northern Spain: Tools to identify the Holocene/Anthropocene transition in coastal environments. *Sci. Total Environ.* 470–471, 240–247. <https://doi.org/10.1016/j.scitotenv.2013.09.064>

Liu, J., Li, J., Lin, T., Liu, D., Xu, Y., Chaemfa, C., Qi, S., Liu, F., Zhang, G., 2013. Diurnal and nocturnal variations of PAHs in the Lhasa atmosphere, Tibetan Plateau: Implication for local sources and the impact of atmospheric degradation processing. *Atmos. Res.* 124, 34–43. <https://doi.org/10.1016/j.atmosres.2012.12.016>

Liu, Y., Beckingham, B., Ruegner, H., Li, Z., Ma, L., Schwientek, M., Xie, H., Zhao, J., Grathwohl, P., 2013. Comparison of Sedimentary PAHs in the Rivers of Ammer (Germany) and Liangtan (China): Differences between Early- and Newly-Industrialized

Countries. *Environ. Sci. Technol.* 47, 701–709. <https://doi.org/10.1021/es3031566>

Lorgeoux, C., Moilleron, R., Gasperi, J., Ayrault, S., Bonté, P., Lefèvre, I., Tassin, B., Bonté, P., Lefèvre, I., Tassin, B., 2016. Temporal trends of persistent organic pollutants in dated sediment cores: Chemical fingerprinting of the anthropogenic impacts in the Seine River basin, Paris. *Sci. Total Environ.* 541, 1355–1363. <https://doi.org/10.1016/j.scitotenv.2015.09.147>

Markiewicz, A., Björklund, K., Eriksson, E., Kalmykova, Y., Strömvall, A.-M., Siopi, A., 2017. Emissions of organic pollutants from traffic and roads: Priority pollutants selection and substance flow analysis. *Sci. Total Environ.* 580, 1162–1174. <https://doi.org/10.1016/j.scitotenv.2016.12.074>

Motelay-Massei, A., Ollivon, D., Garban, B., Chevreuil, M., 2003. Polycyclic aromatic hydrocarbons in bulk deposition at a suburban site: assessment by principal component analysis of the influence of meteorological parameters. *Atmos. Environ.* 37, 3135–3146. [https://doi.org/10.1016/S1352-2310\(03\)00218-8](https://doi.org/10.1016/S1352-2310(03)00218-8)

Motelay-Massei, A., Ollivon, D., Garban, B., Teil, M.J., Blanchard, M., Chevreuil, M., 2004. Distribution and spatial trends of PAHs and PCBs in soils in the Seine River basin, France. *Chemosphere* 55, 555–565. <https://doi.org/10.1016/j.chemosphere.2003.11.054>

Motelay-Massei, A., Ollivon, D., Garban, B., Tiphagne-Larcher, K., Zimmerlin, I., Chevreuil, M., 2007. PAHs in the bulk atmospheric deposition of the Seine river basin: Source identification and apportionment by ratios, multivariate statistical techniques and scanning electron microscopy. *Chemosphere* 67, 312–321. <https://doi.org/10.1016/j.chemosphere.2006.09.074>



- Mouhri, A., Motelay-massei, A., Massei, N., Fournier, M., Laignel, B., 2008. Polycyclic aromatic hydrocarbon transport processes on the scale of a flood event in the rural watershed of Le Bebec, France. *Chemosphere* 73, 443–450. <https://doi.org/10.1016/j.chemosphere.2008.05.046>
- Oliveira, C., Martins, N., Tavares, J., Pio, C., Cerqueira, M., Matos, M., Silva, H., Oliveira, C., Camões, F., 2011. Size distribution of polycyclic aromatic hydrocarbons in a roadway tunnel in Lisbon, Portugal. *Chemosphere* 83, 1588–1596. <https://doi.org/10.1016/j.chemosphere.2011.01.011>
- Ollivon, D., Garban, B., Chesterikoff, A., 1995. Analysis of the Distribution of Some Polycyclic Aromatic Hydrocarbons in Sediments and Suspended Matter in the River Seine (France). *Water, Air Soil Pollut.* 81, 135–152. <https://doi.org/10.1007/BF00477261>
- Pacyna, J.M., Breivik, K., Münch, J., Fudala, J., 2003. European atmospheric emissions of selected persistent organic pollutants, 1970–1995. *Atmos. Environ.* 37, 119–131. [https://doi.org/10.1016/S1352-2310\(03\)00240-1](https://doi.org/10.1016/S1352-2310(03)00240-1)
- Rocher, V., Azimi, S., Gasperi, J., Beuvin, L., Muller, M., Moilleron, R., Chebbo, G., 2004. Hydrocarbons and metals in atmospheric deposition and roof runoff in central Paris. *Water, Air, Soil Pollut.* 159, 67–86. <https://doi.org/10.1023/B:WATE.0000049165.12410.98>
- Rodenburg, L.A., Valle, S.N., Panero, M.A., Muñoz, G.R., Shor, L.M., 2010. Mass Balances on Selected Polycyclic Aromatic Hydrocarbons in the New York–New Jersey Harbor. *J. Environ. Qual.* 39, 642. <https://doi.org/10.2134/jeq2009.0264>

- Rose, N.L., Rippey, B., 2002. The historical record of PAH, PCB, trace metal and fly-ash particle deposition at a remote lake in north-west Scotland. *Environ. Pollut.* 117, 121–132. [https://doi.org/10.1016/S0269-7491\(01\)00149-X](https://doi.org/10.1016/S0269-7491(01)00149-X)
- Schwientek, M., Rügner, H., Beckingham, B., Kuch, B., Grathwohl, P., 2013. Integrated monitoring of particle associated transport of PAHs in contrasting catchments. *Environ. Pollut.* 172, 155–162. <https://doi.org/10.1016/j.envpol.2012.09.004>
- Schwientek, M., Rügner, H., Scherer, U., Rode, M., Grathwohl, P., 2017. A parsimonious approach to estimate PAH concentrations in river sediments of anthropogenically impacted watersheds. *Sci. Total Environ.* 601–602, 636–645. <https://doi.org/10.1016/j.scitotenv.2017.05.208>
- Shi, Z., Tao, S., Pan, B., Liu, W.X., Shen, W.R., 2007. Partitioning and source diagnostics of polycyclic aromatic hydrocarbons in rivers in Tianjin, China. *Environ. Pollut.* 146, 492–500. <https://doi.org/10.1016/j.envpol.2006.07.009>
- Sicre, M.-A., Fernandes, M.B., Pont, D., 2008. Poly-aromatic hydrocarbon (PAH) inputs from the Rhône River to the Mediterranean Sea in relation with the hydrological cycle: Impact of floods. *Mar. Pollut. Bull.* 56, 1935–1942. <https://doi.org/10.1016/j.marpolbul.2008.07.015>
- Sofowote, U.M., Hung, H., Rastogi, A.K., Westgate, J.N., Deluca, P.F., Su, Y., McCarry, B.E., 2011. Assessing the long-range transport of PAH to a sub-Arctic site using positive matrix factorization and potential source contribution function. *Atmos. Environ.* 45, 967–976. <https://doi.org/10.1016/j.atmosenv.2010.11.005>
- Uher, E., Mirande-Bret, C., Gourlay-Francé, C., 2016. Assessing the relation between

anthropogenic pressure and PAH concentrations in surface water in the Seine River basin using multivariate analysis. *Sci. Total Environ.* 557–558, 551–561. <https://doi.org/10.1016/j.scitotenv.2016.03.118>

Van Metre, P.C., Mahler, B.J., Furlong, E.T., 2000. Urban Sprawl Leaves Its PAH Signature. *Environ. Sci. Technol.* 34, 4064–4070. <https://doi.org/10.1021/es991007n>

Walsh, C.J., Roy, A.H., Feminella, J.W., Cottingham, P.D., Groffman, P.M., Morgan, R.P., 2005. The urban stream syndrome: current knowledge and the search for a cure. *J. North Am. Benthol. Soc.* 24, 706–723. <https://doi.org/10.1899/04-028.1>

Yunker, M.B., Macdonald, R.W., Fowler, B.R., Cretney, W.J., Dallimore, S.R., McLaughlin, F.A., 1991. Geochemistry and fluxes of hydrocarbons to the Beaufort Sea shelf: A multivariate comparison of fluvial inputs and coastal erosion of peat using principal components analysis. *Geochim. Cosmochim. Acta* 55, 255–273. [https://doi.org/10.1016/0016-7037\(91\)90416-3](https://doi.org/10.1016/0016-7037(91)90416-3)

Table captions:

Table 1: Comparison of total PAH atmospheric fallout in the Orge River catchment (this study) with those values reported in the literature

Table 2: Comparison of Orge River PAH contamination in the particulate and dissolved phases (this study) with those values reported in previous studies conducted in France and Europe

Table 3: Comparison of subcatchment contributions to water discharge at the outlet and particulate PAH export with respect to their respective surfaces

Table 4: Specific particulate fluxes of PAH in  $\text{g km}^{-2} \text{y}^{-1}$  and **specific** sediment yields in  $\text{t km}^{-2} \text{y}^{-1}$  for contrasting catchments across the world

Table 1: Atmospheric deposition under each land use and corresponding PAH stock in the Orge River catchment

Table1

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Location	Type	Atmospheric deposition ( $\mu\text{g m}^{-2} \text{y}^{-1}$ )		Reference
Orsay - France	Peri-urban	182	$\Sigma$ 13PAHs	This study
Paris - France	Urban	57 - 472	$\Sigma$ 14PAHs	(Garban et al., 2002)
	Urban	119		(Blanchard et al., 2007)
Créteil (Paris suburb) - France	Urban	153		(Azimi et al., 2005)
Orgeval - France	Rural	157	$\Sigma$ 13PAHs	(Gateuille et al., 2014a)
Le Havre - France	Urban	70		(Motelay-Massei et al., 2003)
Southern Germany	Rural	200	$\Sigma$ 17PAHs	(Gocht et al., 2007)
Northern Spain	Rural (National Park)	14 - 189	$\Sigma$ 14PAHs	(Foan et al., 2015)

Table2

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Location	Location	PAH concentration in SPM $\mu\text{g g}^{-1}$	PAH concentration in the dissolved phase $\text{ng L}^{-1}$		References
<b>Orge River (France)</b>	Peri-urban/Urban	4.6 (1-45.2)	41 (11-221)	$\Sigma 13\text{PAHs}$	This study
<b>Orgeval River (France)</b>	Rural	4.0 (0.05-14)	72 (60-112)	$\Sigma 13\text{PAHs}$	(Gateuille et al., 2014a)
<b>Seine River (France)</b>	Urban	5.8 (2.8-8.7)	80 (26-166)	$\Sigma 16\text{PAHs}$	(Ollivon et al., 1995)
<b>Moselle River (France)</b>	Industrial	4.5 (2.3-7.7)	-	$\Sigma 16\text{PAHs}$	(Le Meur et al., 2017)
<b>Rhône River (France)</b>	Urban	2.3 (0.02-7)	11 (3-89)	$\Sigma 14\text{PAHs}$	(Sicre et al., 2008)
<b>Körsch River (Germany)</b>	Urban	4.6	29	$\Sigma 15\text{PAHs}$	(Schwientek et al., 2013)
<b>Elbe River (Germany)</b>	Urban	4.7 (2.5-8.9)	-	$\Sigma 16\text{PAHs}$	(Heemken et al., 2000)
<b>Mackenzie River (Canada)</b>	Rural	$2.1 \pm 0.4$	-	$\Sigma 16\text{PAHs}$	(Yunker et al., 1991)

Table3

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<b>Subcatchments</b>	<b>Egly</b>	<b>Yvette</b>	<b>Viry<sub>sub</sub></b>	
<b>Proportion of the total catchment surface (%)</b>	50	20	30	
<b>Proportion of the total water discharge (%)</b>	52 ± 3	28 ± 3	21 ± 4	
<b>Specific sediment yield (t km<sup>-2</sup> y<sup>-1</sup>)</b>	<b>4.2</b>	<b>7.1</b>	<b>3.1</b>	
<b>Proportion of the total sediment load (%)</b>	<b>48</b>	<b>30</b>	<b>22</b>	
<b>Particulate PAH export by each subcatchment in g d<sup>-1</sup> (proportion of the total export in %)</b>	January 2016	9.9 (32%)	1.5 (5%)	18.9 (62%)
	April 2016	14.4 (32%)	6.9 (15%)	23.9 (53%)
	August 2016	7.0 (111%)	3.4 (54%)	- 4.0 (- 65%)
	November 2016	5.4 (19%)	5.2 (19%)	17.0 (62%)

Table4

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Location		Type	PAH specific load (g km <sup>-2</sup> y <sup>-1</sup> )	Specific sediment yield (t km <sup>-2</sup> y <sup>-1</sup> )	Ratio PAH specific load/specific sediment yield (g t <sup>-1</sup> )	Reference
<b>Orge River catchment (France)</b>	Dourdan	Rural	10.3 ± 1.1	7.4	1.4	This study
	Egly	Rural/Suburban	19.1 ± 1.0	4.2	4.5	
	Yvette	Rural/Suburban	24.8 ± 0.7	7.1	3.5	
	Viry <sub>sub</sub>	Urban	46.7 ± 2.5	3.1	15.1	
	Viry (outlet)	Urban	28.7 ± 0.1	4.4	6.5	
<b>Sucy (France)</b>		Urban	1200	70	17	(Hannouche et al., 2017)
<b>Orgeval River catchment (France)</b>		Rural	0.2-30.4	18.9	1.6	(Gateuille et al., 2014a)
<b>Seine River catchment (France)</b>		Urban	70	12*	5.8	(Ollivon et al., 1995) (*Delmas et al., 2012)
<b>Moselle River (France)</b>		Urban/industrial	13.1	2.9	4.5	(Le Meur et al., 2017)
<b>Rhône River (France)</b>		Mixed	170	76	2.2	(Sicre et al., 2008)
<b>Körsh River (Germany)</b>		Urban	161	35	4.6	(Schwientek et al., 2013)
<b>Elbe River (Germany)</b>		Urban	28.5	5.3	5.4	(Heemken et al., 2000)
<b>Mackenzie River (Canada)</b>		Rural (Forest)	27	29	0.7	(Yunker et al., 1991)



Table5

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	<b>Agricultural soils</b>	<b>Forest soils</b>	<b>Urban soils</b>
<b>Atmospheric deposition (kg y<sup>-1</sup>)</b>	74 ± 10	53 ± 10	46 ± 10
<b>PAH stock (tons)</b>	840	75	309
<b>(min-max)</b>	(14-4511)	(13-138)	(7-2801)

### Figure captions

Figure 1: Land uses in the Orge River catchment (source: Corine Land Cover, 2012) and locations of atmospheric fallout and river monitoring stations.

Figure 2: River flow and precipitation recorded in the Orge River catchment from January to December 2016. Low flow periods (orange): August 2016 (08/24/2016-08/29/2016). Average discharge conditions (green): January 2016 (01/21/2016-01/25/2016), November 2016 (11/16/2016-11/22/2016) and December 2016 (12/05/2016, 12/15/2016 and 01/03/2017). High flow period (blue): April 2016 (04/07/2016-04/11/2016). **Extreme flood (black): June 2016 (06/06/2016: water flow decrease).**

Figure 3: Spatial and temporal variations of dissolved PAH proportions (a, b), temporal variations of PAH dissolved concentrations in ng L<sup>-1</sup> (c) and spatial variation of particulate contents in µg g<sup>-1</sup> (d). Significant differences between groups (sites or campaigns) are represented by (\*)

Figure 4: Riverine export of PAH in the dissolved phase (a) and the particulate phase (b) for different sampling campaigns conducted in the Orge River in 2016

Figure 1: Annual mass balance of total PAH fluxes (dissolved and particulate) estimated for the Orge River catchment and the subcatchments of Egly, Yvette, Dourdan and Virysub associated with PAH atmospheric deposition and the annual accumulation rate for each subcatchment

Figure1

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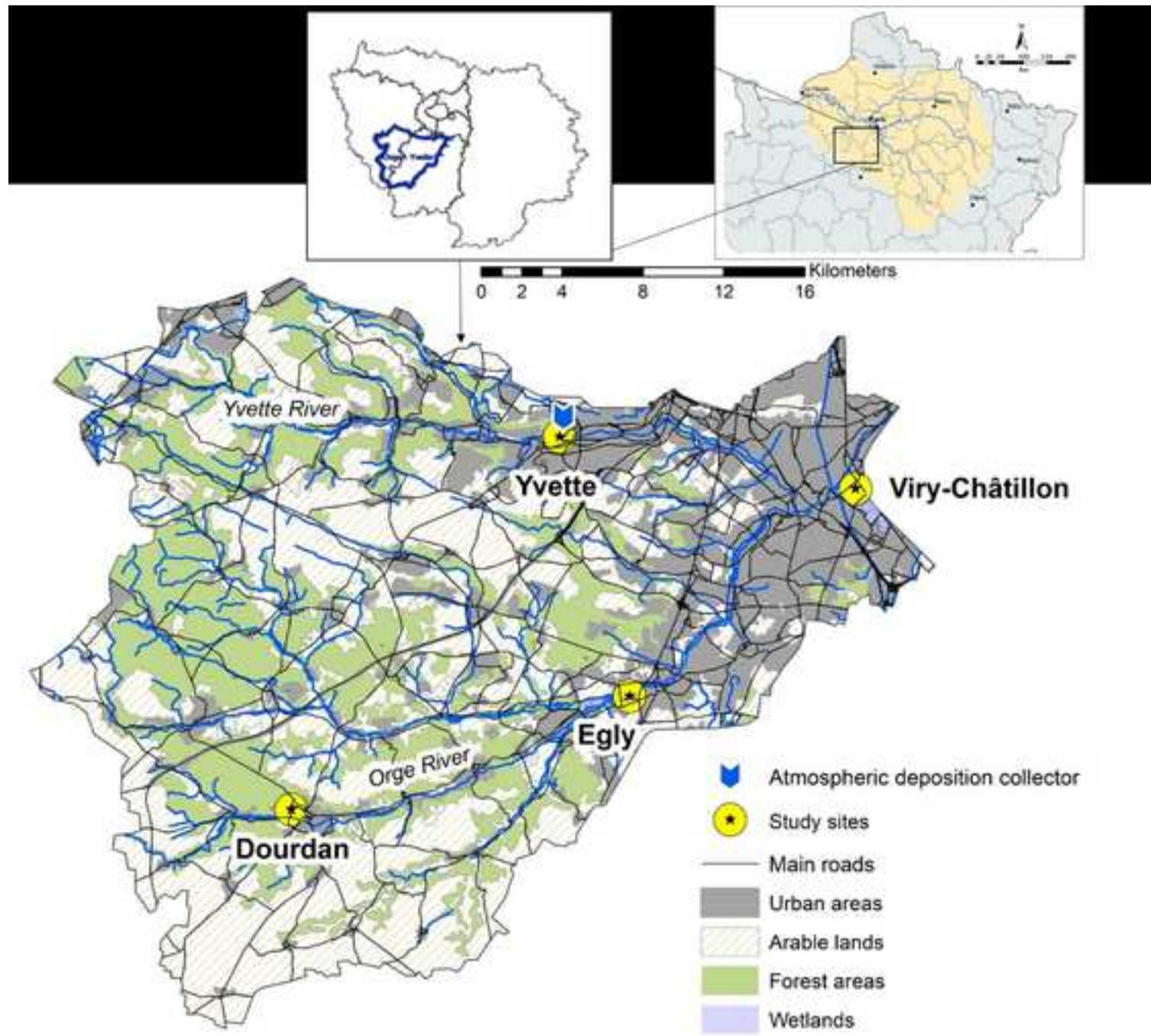


Figure2

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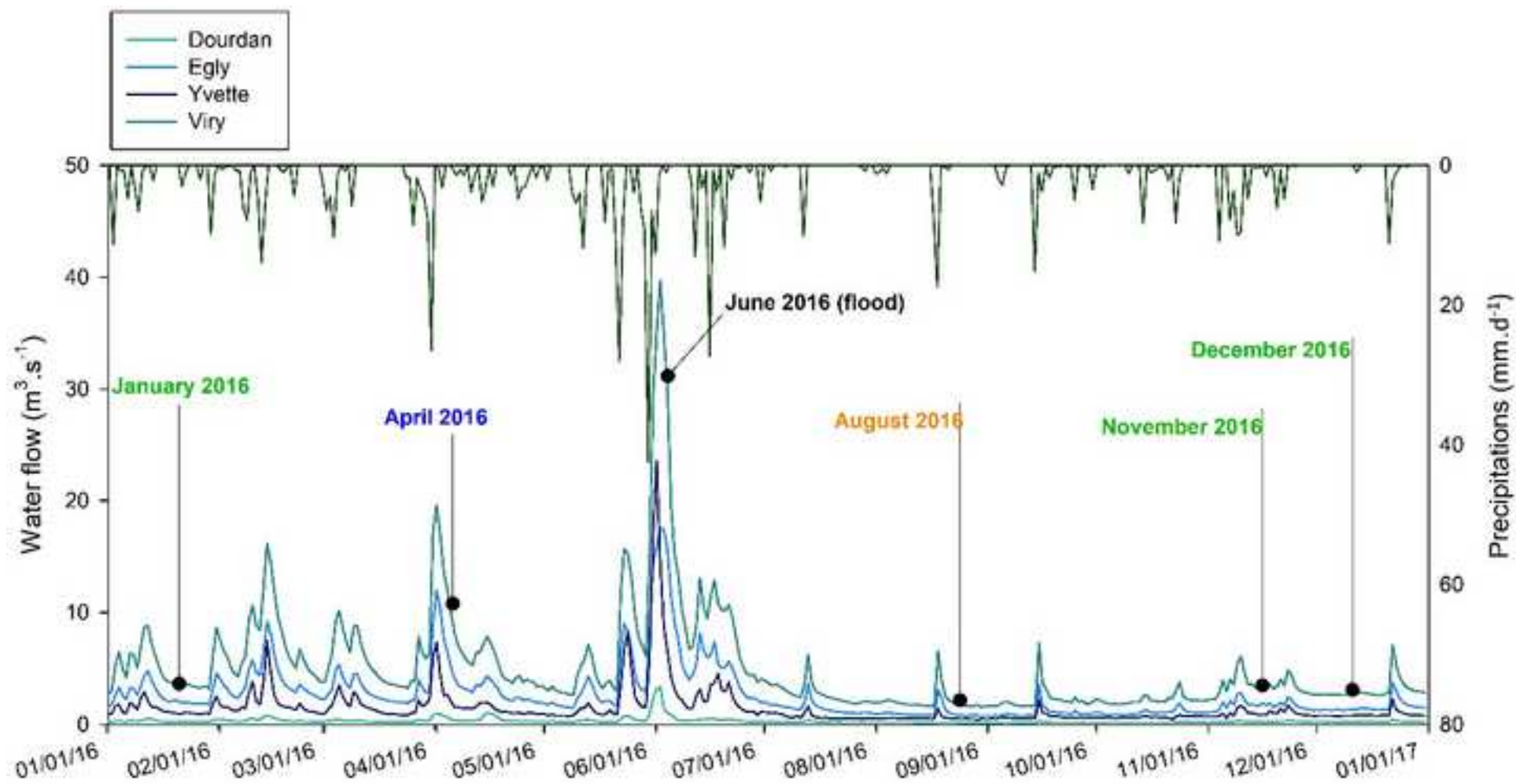


Figure3

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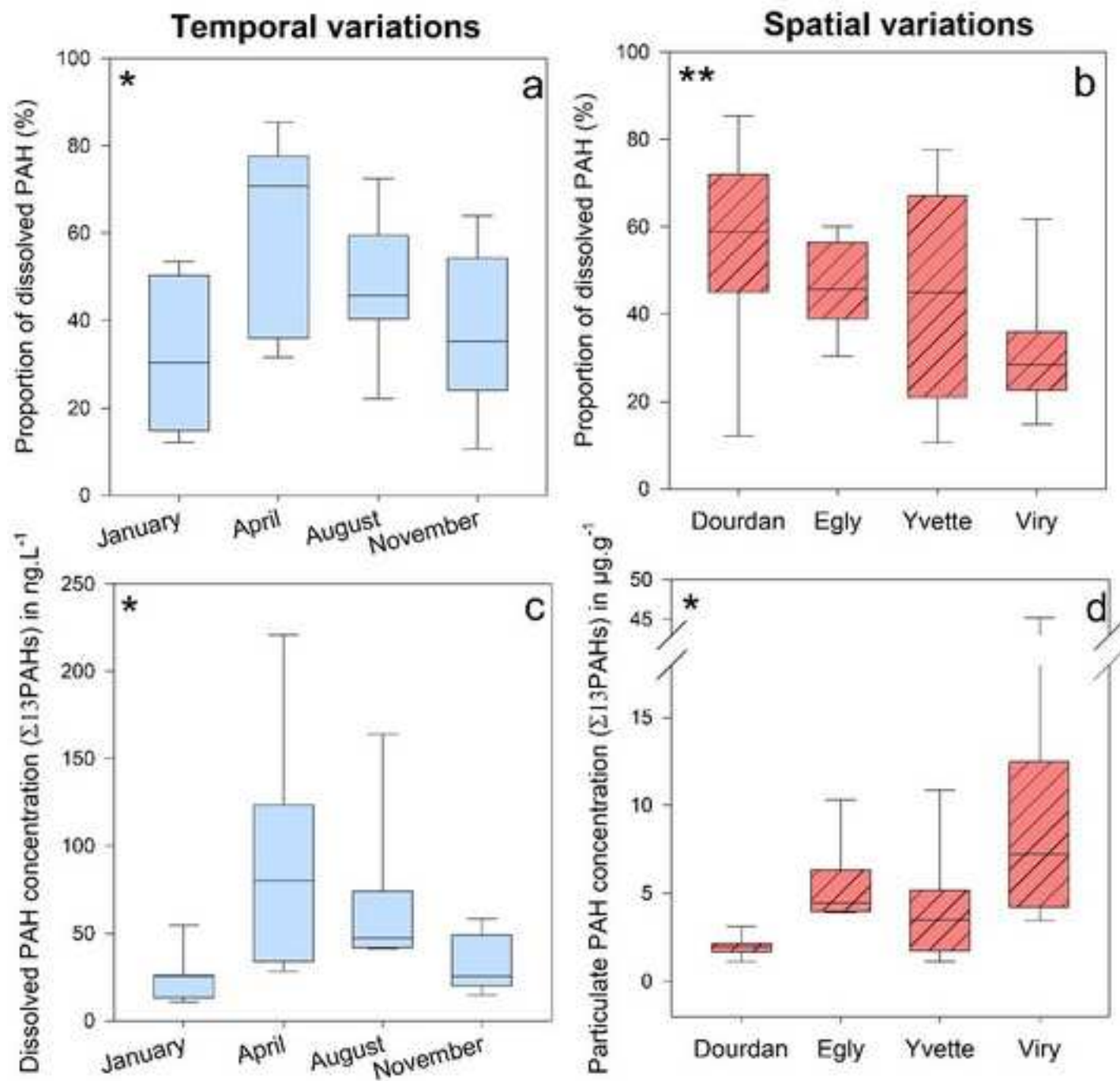


Figure4

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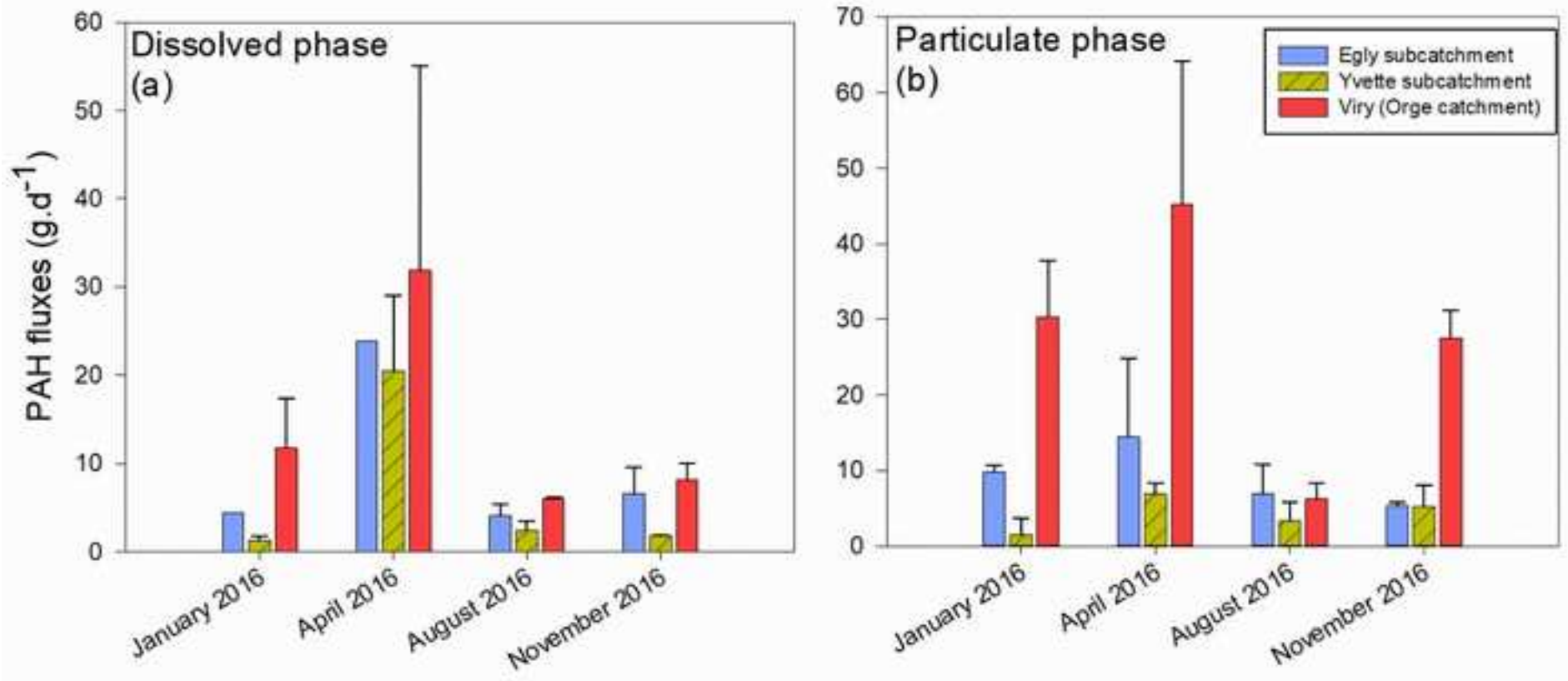
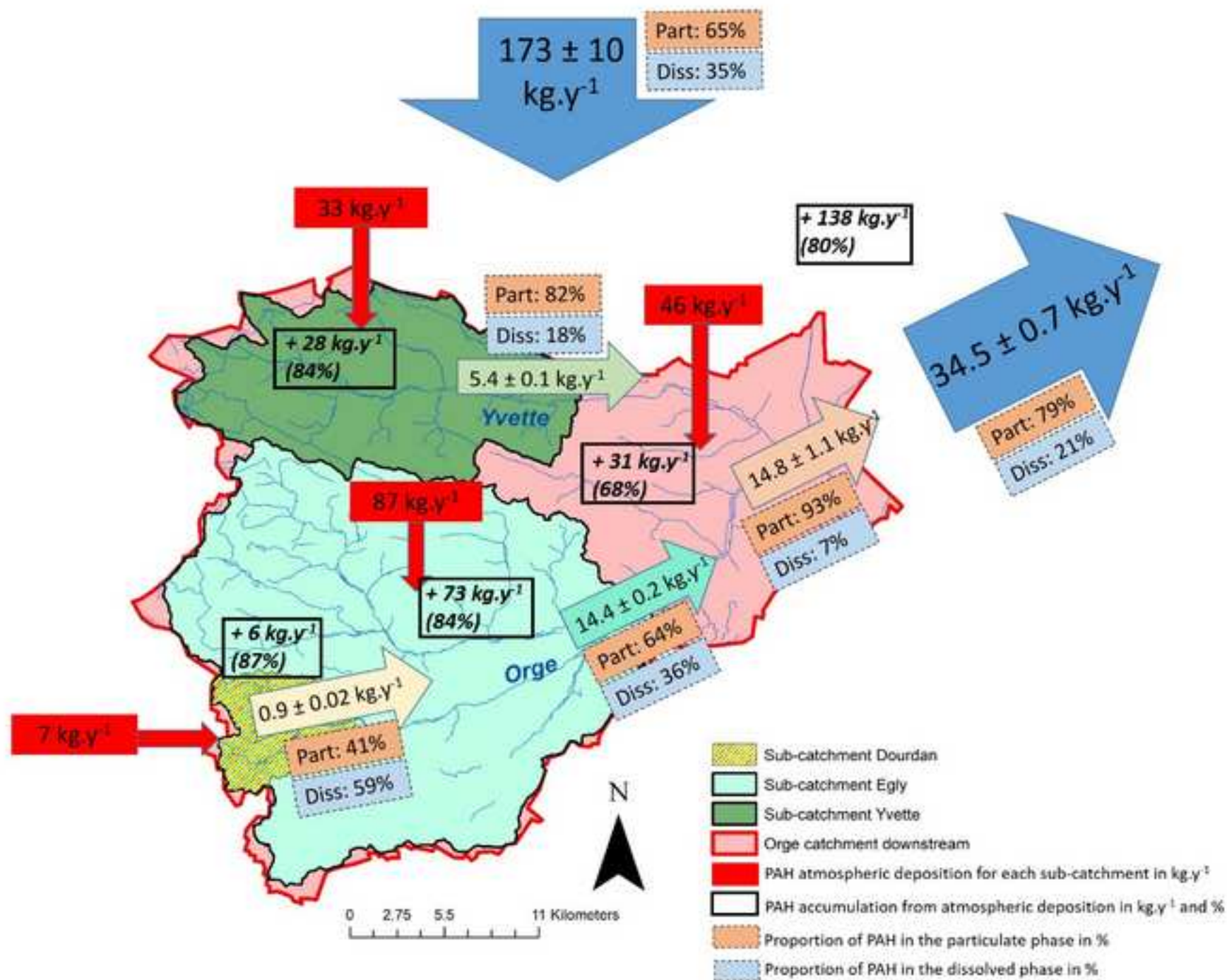


Figure5

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