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1 **Record of pharmaceutical products in river sediments: a powerful tool to**
2 **assess the environmental impact of urban management?**

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16 **Abstract:**

17 The occurrence of eight pharmaceutical products (PPs) from various therapeutic classes was
18 studied in urban sediments collected upstream a small dam in Orleans, France. Since PPs are
19 globally distributed contaminants since the 1950s, their spatial and historical distribution was
20 documented in order to better understand the impact of recent urban management on the
21 chemical quality of sediments. Concentrations of the PPs mainly ranged between 1 and 10 ng
22 g⁻¹ within the two cores. The chronology of core LOI13-2, based on radionuclide analysis
23 (¹³⁷Cs and ²¹⁰Pb), enabled the characterization of the changes or shift in PPs over the last 50
24 years, impacted by the effluent deflections i) of a large wastewater treatment plant in 1989,
25 and ii) of 3 smaller wastewater treatment plants between 2003 and 2009. In most cases the
26 deepest occurrence of each PP in the core matched the market authorization date of different

27 pharmaceuticals, indicating that some PPs can be used as chronomarkers. The study reveals
28 that recent management of effluent discharges within the watershed improved the chemical
29 quality of these sediments. In view of the persistence of PP pollution in trapped sediments, a
30 total stock of 763 ± 565 g for the selected PPs was estimated at the millstream scale..
31 Superficial sediments therefore represent a potential source of downstream pollution in the
32 event of the removal of a small existing dam.

33 **Keywords:** Urban Dam; Sediments; Pharmaceutical Products; Anthropogenic
34 Management; Wastewater Treatment Plant

35

36 **1. Introduction**

37 The growing use of pharmaceutical products (PPs) for human and animal therapies
38 since the 1950s raises concerns about the environmental fate of these products and their
39 metabolites within the aquatic environment because of their non-negligible excretion via urine
40 and faeces (Daughton and Ternes, 1999; Lienert et al., 2007; Farré et al., 2008). Moreover,
41 because they are exclusively generated by human activities, PPs could provide unique
42 stratigraphic markers to allow the assessment and dating of anthropogenic impacts on the
43 environment in a similar manner to how microplastics, synthetic fibers or pesticides have
44 been used (Waters et al., 2016; Zalasiewicz et al., 2016). Nevertheless, the occurrence of PPs
45 is under reported within sedimentary archives (Bernhardt et al., 2017), despite their significant
46 concentrations in numerous water compartments. The question about their long-term fate also
47 represents a challenge for both an effective water resource management (Loos et al., 2009;
48 Mompelat et al., 2009; Petrie et al., 2015) and a better tracking of the impacts of human
49 activities on the environment.

50 Pharmaceutical products contain a wide variety of organic compounds (i.e. more than
51 3000 in France according to the French National Agency for Medicines and Health Products
52 Safety, ANSM, 2014). These pharmaceuticals are designed to prevent and treat various
53 diseases and to improve health in general. The main sources of PPs are domestic, hospital and
54 industrial wastewaters (specifically pharmaceutical plants) that reach water bodies through
55 sewage systems (Fick et al., 2009; Verlicchi et al., 2010) due to the inefficiency of classical or
56 advanced wastewater treatment plants (WTP) to totally remove PPs (Busetti et al., 2015;
57 Petrie et al., 2015). Accordingly, PPs enter the natural environment and potentially cause
58 severe ecological issues and/or affect human health. Widespread contamination has been
59 observed in surface, marine and ground waters (e.g. Heberer, 2002; McEneff et al., 2014;
60 Lopez et al., 2015) as well as soils and sediments (e.g. Silva et al., 2011; Tamtam et al., 2011;
61 Vazquez-Roig et al., 2012). Furthermore, the early studies related to the ecotoxicity of PPs at
62 field-relevant concentrations have demonstrated that these pollutants are harmful to both
63 invertebrates and vertebrates, even in low or natural concentrations (Brodin et al., 2013; De
64 Castro-Català et al., 2016, Väitalo et al., 2017).

65 Beyond their hydrological impact on rivers (Hubbard et al., 2016), WTPs contribute a
66 constant discharge of PPs in natural waters. Nevertheless, this flux is not random. For
67 example, Klaminder et al. (2015) demonstrated a good correlation between the effluent and
68 the downstream sediment concentrations of oxazepam (an anxiolytic drug), indicating that the
69 impact of WTPs on the environment can be monitored (i) by the effluent concentration, and
70 (ii) since the market authorization date of each compound.

71 The occurrence of PPs is closely related to the worldwide increase in population and
72 the ongoing urbanization since World War II and therefore represents a major environmental
73 issue for urban ecosystems (Heberer, 2002; Taylor and Owens, 2009). While numerous
74 studies have been published concerning the occurrence of PPs in natural waters, there is still a

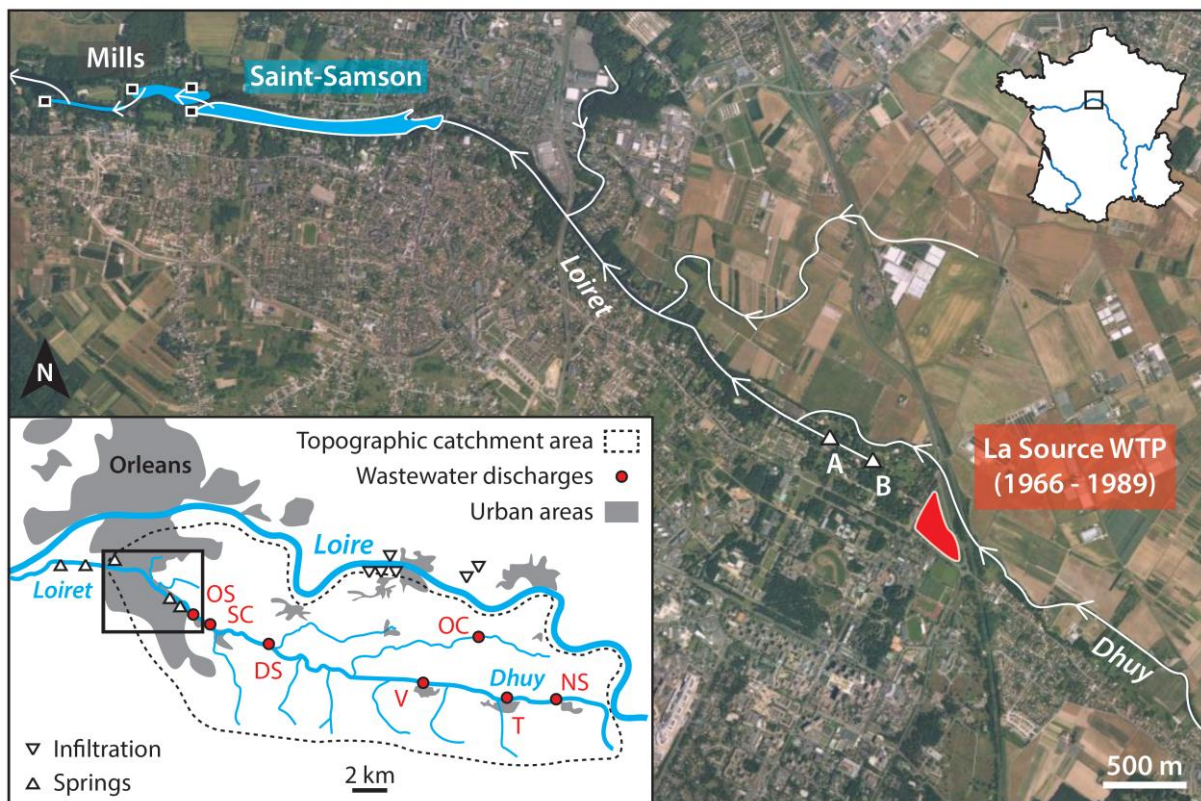
75 lack of research on the spatial and temporal trends of this widespread aquatic pollution in
76 sediments (Petrie et al., 2015). Sediments are however a relevant compartment for this type of
77 tracking thanks to their ability to record the evolution of contamination over several decades
78 and beyond (Tamtam et al., 2011; Dubois and Jacob, 2016; Lorgeoux et al., 2016).
79 Furthermore, PPs can be considered as very reactive contaminants, exhibiting some affinity
80 for both organic and inorganic surfaces (Stein et al., 2008; Zhou and Broodbank, 2014;
81 Thiebault et al., 2016a, 2016b). Moreover, several PPs have recently been added to the list of
82 priority substances in the amended Water Framework Directive of the European Union
83 (European Commission, 2013), indicating that PPs are now considered as a potential threat for
84 various organisms, including humans (De Jongh et al., 2012). Finally, an assessment of the
85 stock of contaminated sediments accumulated in small ponds and reservoirs should be
86 considered since the Water Framework Directive (European Commission, 2000) has
87 recommended the removal of dams without any practical function in order to improve the
88 ecological continuity of superficial water bodies. Such an assessment is required to prevent
89 the potential downstream propagation of PPs induced by physical modifications within the
90 reservoir such as dredging or removal, or by chemical changes in the water column.

91 In view of the potential hazards of these PPs and the need to acquire data to assess and
92 date the impact of anthropogenic activities on the environment, eight PPs were selected
93 among the most frequently detected compounds in EU waters (Tamtam et al., 2008; Loos et
94 al., 2009; Joigneaux, 2011), namely acetaminophen, atenolol, codeine, metoprolol, oxazepam,
95 ofloxacin, trimethoprim and sulfamethoxazole. In the study presented here, the presence of
96 PPs in sediments collected in a small millstream located in the urban area of Orleans, France,
97 was investigated. This work aimed to (i) quantify PPs within a sedimentary archive to assess
98 their potential as new chronomarkers; (ii) establish the long-term chronology of PP
99 contamination related to the evolution of WTPs in the catchment in order to evaluate the link

100 between the discharge of these contaminants and sedimentary contamination; (iii) observe the
101 spatial variation in the occurrence of PPs in two cores to assess the contamination at a
102 reservoir scale..

103 2. Environmental settings

104 2.1. Hydrological context



105
106 Figure 1. Aerial photograph showing the location of the Saint-Samson millstream within the urban
107 area of Orleans (www.geoportail.gouv.fr). White lines indicate rivers. A = Les Abîmes spring; B = Le
108 Bouillon spring. The inset illustrates the hydrographic context of the Loire karst upstream of Orleans,
109 with groundwater circulation between infiltrations and springs. The effluent inputs from WTPs to the
110 Dhuy River are marked by red dots: OS = Orleans-la-Source - SC = Saint-Cyr-en-Val - DS = Darvoy
111 and Sandillon - OC = Ouvrouer-les-Champs - V = Vienne-en-Val - T = Tigy - NS = Neuvy-en-Sullias
112 (see Table 1).

113 The Saint-Samson millstream is a 0.125 km² pond located in the southern part of the
114 Orleans urban area in France (Figure 1). This area is characterized by a succession of
115 watermills built by monks during the medieval period along the Loiret river. Among them, the
116 upstream mill of Saint-Samson maintains an elongated pond at an elevation of 90.87 m a.s.l.
117 with a maximum length of 1.3 km and a mean width of 80 m. Its outflow and water level are
118 currently regulated by a small dam. It is fed by the Loiret river, whose waters come from
119 springs connected to the fluvial-karstic system of the Loire river (Albéric and Lepiller, 1998;
120 Albéric, 2004), and by the Dhuy river, a 30 km long tributary that flows in the agricultural
121 lowlands of the Val d'Orléans (Figure 1). A recent report revealed that the origin of
122 suspended sediments in the Saint-Samson millstream can be divided between the Dhuy river
123 (216 km²), the urban catchment (34 km²) and, for a negligible fraction, the karst (Geo-hyd,
124 2009). This report also indicated that during winter high-flows, the Dhuy river is the main
125 sediment supplier for the Loiret river.

126 At the confluence with the Loiret river, the mean annual flow of the Dhuy river is
127 about 0.7 m³.s⁻¹ with minimum values during the end of summer (0.07 m³.s⁻¹) and maximum
128 values during winter (1.49 m³.s⁻¹) (source: www.hydroeau.france.fr). With such weak flows, a
129 permanent water table is maintained throughout the year due to numerous anthropogenic
130 developments along the river banks such as small dikes, platforms, walls and wooden studs.
131 Eutrophication has led to the proliferation of filamentous green algae (*Egeria densa* and
132 *Elodea nuttallii*), regularly observed during summer over the last 40 years and was one of the
133 main reasons for the deflection of effluent discharges from the millstream.

134

135 **2.2. Sanitation systems evolution**

136 Land-use evolution since the 1950s resulted in the expansion of urban areas, mainly
137 concentrated in the southern part of Orleans. Both the growing urbanization and the evolution
138 of sanitation techniques led to the construction of several WTPs between 1966 and 2002
139 (Table 1 and Figure 1). The largest WTP, located close to the confluence between the two
140 rivers in Orléans-La-Source (OS), was built in 1966 to support a population-equivalent (PE)
141 of 60,000 (Figure 1). It discharged wastewaters into the Dhuy river until 1989. After this date
142 a diversion via pipelines downstream to the Loire river was implemented to limit the
143 eutrophication of the Loiret river. Meanwhile, other plants were constructed in the upstream
144 catchment over the last 40 years to support a PE of < 15,000 (Table 1). However, several
145 alterations were recently made to some of these plants to comply with current regulations and
146 extend sanitation access. The WTP of Saint-Cyr-en-Val (SC) was connected to OS in 2009
147 while the plants in Darvoy (D) and Sandillon (S) were combined in 2003 to reach a capacity
148 of 8,800 PE (Table 1). Since then, the effluent waters of this broad new sanitation system
149 have flowed directly into the Loire river. Lastly, the capacity of the plant in Neuvy-en-Sullias
150 (NS, Figure 1) was doubled in 2006 from 500 to 1,100 PE, but associated wastewater
151 effluents still feed the Dhuy river (Table 1). It should be mentioned that all the WTPs that
152 discharge their effluent into the millstream are conventional activated sludge treatment plants
153 and provide only incomplete removal of PPs (Verlicchi et al., 2012; Petrie et al., 2015).

154 Table 1. Evolution of wastewater treatment plants along the Dhuy catchment area (sources:
155 SAGE Loiret and <http://assainissement.developpement-durable.gouv.fr/>). TSS: Total Suspended
156 Sediment. Data in brackets indicate the functional parameters of the WTP after renovation and/or
157 deflection. The WTPs of Darvoy and Sandillon had the same discharge point in the Dhuy river and
158 therefore the same distance from the Saint-Samson millstream (red dots on Figure 1). Note that these
159 plants were upgraded and combined in 2003.

Station name	Installation date	Deflection date	Capacity (10 ³ PE)	Flow (m ³ .day ⁻¹)	TSS (kg.day ⁻¹)	Discharge-millstream distance (km)
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Orleans-La-Source	1966	1989	60 (93.9)	n.d. (4050)	n.d. (1072)	4.5
Saint-Cyr-en-Val	1982	2009	2	420	86	7
Darvoy	1987	2003	1	(8.8)	300	94
Sandillon	1988		2		300	85
Vienne-en-Val	1998	-	1.7	400	4	21.5
Ouvrouer-les-Champs	2002	-	0.35	52	2	25
Tigy	1999	-	2	460	9	26
Neuvy-en-Sullias	1978 (2006)	-	0.5 (1.1)	75 (205)	30	29.5

160

161 **3. Materials and methods**

162 **3.1. Acoustic imaging and sample collection**

163 The bathymetry of the millstream was assessed by combining geophysical mapping
164 techniques and handheld measures. A Knudsen® lake bottom profiler was used for seismic
165 surveys in the central part of the millstream at a frequency of 200 kHz. A P-wave velocity of
166 1500 m.s⁻¹ was used to convert the acoustic wave propagation from two-way travel time (in
167 seconds) to water-depth (in meters). A round-trip navigation was carried out with one straight
168 line in the axis of the river followed by a zigzag reaching the shores. To complete this dataset,
169 additional handheld measures of bathymetry and total sediment thickness were performed
170 using a graded stick. The bathymetric map of the millstream as well as the calculation of
171 volume and thickness of sedimentary units were performed using a GIS. Five sedimentary
172 cores were collected along a longitudinal transect using a gravity corer. The two extremes are
173 cores LOI13-2 and LOI13-6, situated at the head of the millstream, and near the outlet of the
174 Saint-Samson basin, respectively.

175 **3.2. Sediment analyses and dating**

176 A multi-proxy approach was adopted to characterize sediments. For each core,
177 magnetic susceptibility (MS) and spectrophotometry were performed every half-centimeter
178 using a Bartington MS2E Sensor and a Konica Minolta CM-700d, respectively. First-

179 derivative spectra (FDS) representative of sediment composition were then calculated for the
180 light spectrum (400-700 nm) and values were represented by color changes within spectral
181 maps (Debret et al., 2011). Core LOI13-2 was sampled at regular intervals (5 cm) for
182 destructive analyses of grain size using a Mastersizer 3000 and for Total Organic Carbon
183 (TOC) using a Rock-Eval 6 (Behar et al., 2001). Dry bulk density was also estimated every 10
184 cm.

185 ^{210}Pb and ^{137}Cs activities were determined in core LOI13-2 to determine the
186 sedimentation rate in the core that we used to provide the time frame for the sedimentary
187 record. Between 6.1 and 11.2 g of dry sample (equivalent to 3-cm thick sections) were placed
188 in counting boxes that were sealed to prevent any loss of ^{222}Rn . The quantification of ^{210}Pb ,
189 ^{226}Ra and ^{137}Cs activities in the sediment samples was performed at the LAFARA
190 underground laboratory (van Beek et al., 2013) using a semi-planar germanium detector (183
191 cm^3 ; PROFILE-FX series; ORTEC/AMETEK). The detector is protected from cosmic
192 radiation by 85 m of rock, ensuring a very low background to the facility. ^{137}Cs and ^{210}Pb
193 activities were determined using the 662 keV line and the 46.5 keV line, respectively. The
194 ^{226}Ra activity was determined using the lines of ^{214}Pb (295 keV and 352 keV) and ^{214}Bi (609
195 keV). Excess ^{210}Pb activities ($^{210}\text{Pb}_{\text{ex}}$) were then determined by correcting the ^{210}Pb activities
196 for the ^{210}Pb supported by ^{226}Ra . The detector is calibrated for natural radionuclides using
197 RGU-1 and RGTh-1 standards. The IAEA Standard #375 was used to determine the detection
198 efficiency for ^{137}Cs .

199 **3.3. Quantification of pharmaceuticals**

200 The standards for the eight investigated PPs, namely acetaminophen, atenolol,
201 codeine, metoprolol, ofloxacin, oxazepam, sulfamethoxazole and trimethoprim were
202 purchased from Sigma-Aldrich assuming a purity of up to 98%. Extraction and separation

203 solvents, dichloromethane (DCM), methanol (MeOH) and acetonitrile (AcN) were purchased
204 from Fisher-Scientific, assuming an analytical grade (purity up to 99.95 %)

205 Sediments were sampled on cores LOI13-2 and LOI13-6 following a regular sampling
206 step (5 cm). They were then dried at 60°C during 48 h before being crushed in a mortar.
207 Around 2 ± 0.1 g of material was afterward extracted by pressurized liquid extraction, using
208 ASE-200 (Dionex). The extraction mixture was MeOH/H₂O (1:1 v/v) and the operating
209 temperature and pressure were 100°C and 1000 psi respectively. Extracts were then dried at
210 60°C and stored at 3°C before liquid chromatography analysis. Samples, standards and
211 controls were prepared with surface waters sampled in the Loiret river (filtered at 0.22 µm),
212 which after analysis were found to be deprived of PPs. The analytical and quantification
213 procedure is further detailed in the supplementary data.

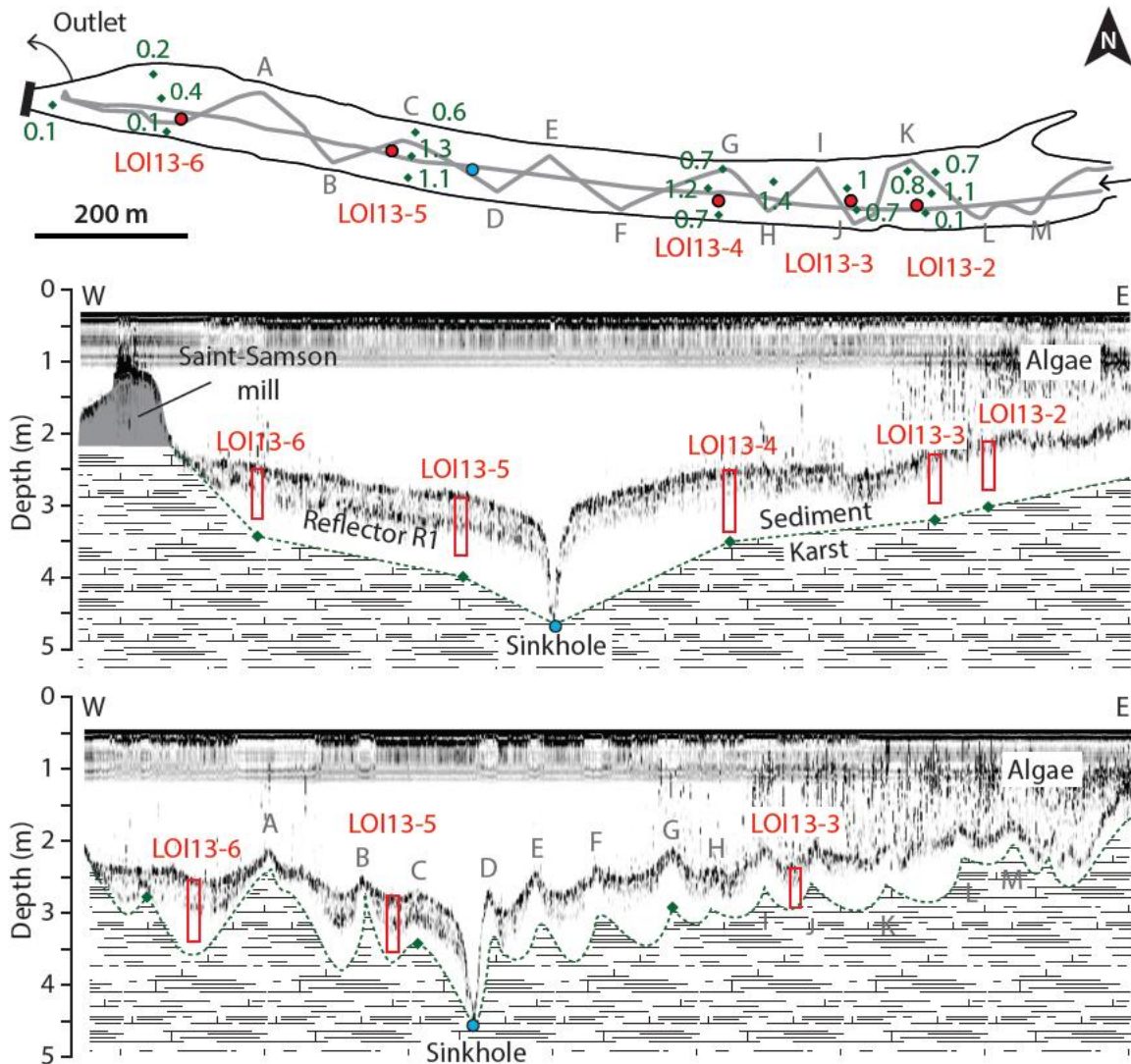
214

215 **4. Results**

216 **4.1. Bathymetry and sediment geometry.**

217 The water-sediment interface can be seen on acoustic images with an uppermost
218 continuous thick reflector throughout the millstream (Figure 2). In the center part, the water
219 level had an overall flat geometry between 2 and 3 m. A circular depression in the water level
220 reaching 4.5 m can also be seen and is likely related to a stream in this karstic context. The
221 acoustic images also displayed a second thick reflector roughly parallel to the river bed
222 (Figure 2). This reflector, labeled R1, was mapped to later calculate the thickness of this top-
223 most acoustic unit U1 across the millstream (Figure S1). Results showed a plate geometry
224 with maximum values fluctuating between 0.3 and 0.5 m thick on the longitudinal axis for an
225 estimated volume of $21,000 \pm 100$ m³. This value represents almost a fifth of the total
226 sediment stock (estimated to $102,000 \pm 6,600$ m³) trapped in the Saint-Samson millstream

227 according to the handheld measures and volume calculations performed by a design office in
 228 2006 (Geo-hyd, 2009).



229
 230 Figure 2. Navigation grid (grey lines), core locations (red dots), punctual sediment thickness measures
 231 (green dots) and acoustic images (200 kHz) of sedimentary infills within the Saint-Samson millstream.
 232 Grey letters refer to angles in the navigation grid. The dashed green line is an estimation of the karst-
 233 sediment interface depth based on measures of total sediment thickness.

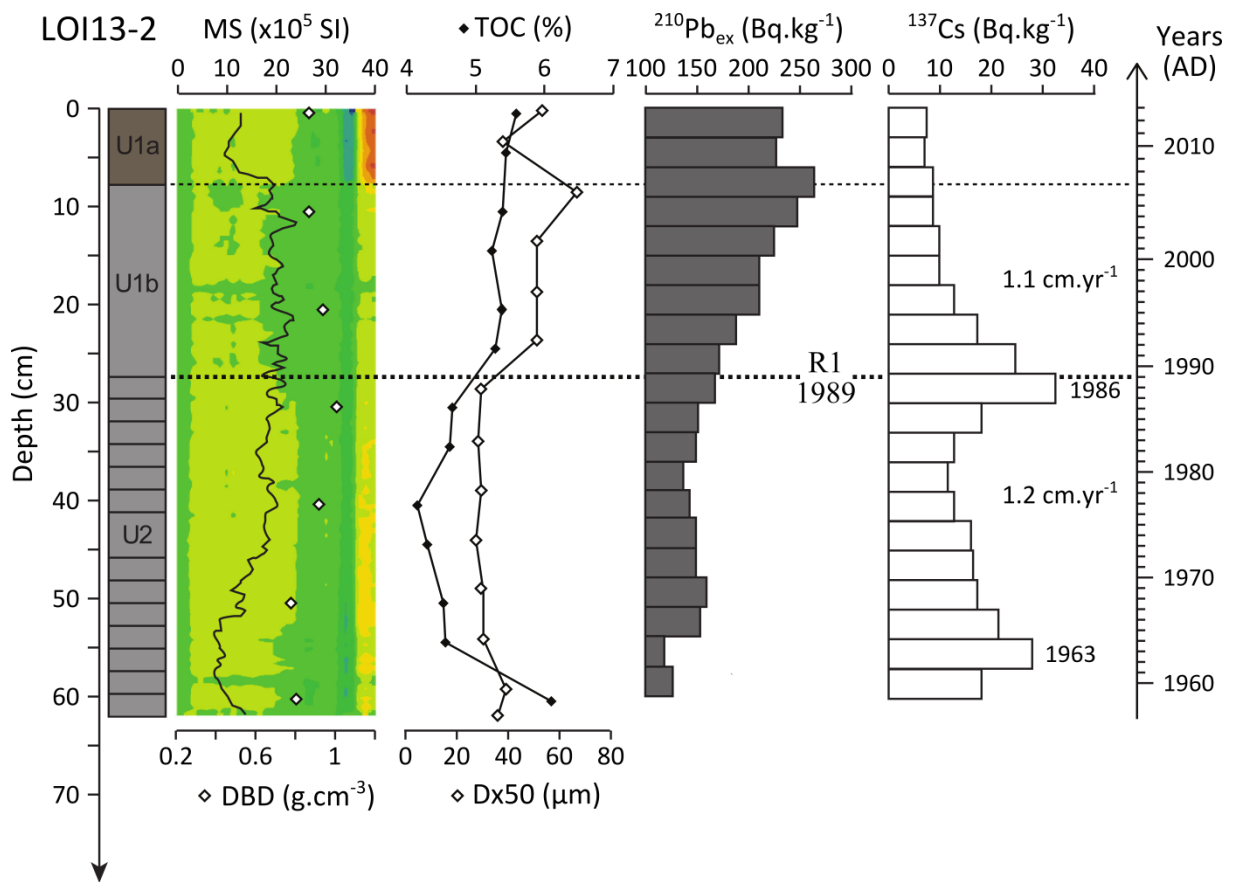
234 **4.2. Description of the sediment**

235 Within the five collected cores (maximum length: 90 cm), the sediment is made of a
 236 greyish to brownish mud rich in water and in small leaf remains. Three sedimentary units

237 have been distinguished and correlated core-to-core on the basis of visual observations,
238 spectral maps and MS values (Figures 3 and S2). The topmost brownish unit, labeled U1a, 10
239 cm thick, was characterized by elevated values between d675 and d695 (yellow to orange
240 strips on the right side of the spectral map in Figure 3). The second unit, labeled U1b, was
241 made of a greyish sediment with a thickness ranging from 20 cm in cores LOI13-2 and
242 LOI13-6 to 41 cm in core LOI13-3. The left side of the spectral map indicates low values
243 fluctuating from d405 to d545 across this unit. However, this spectral signature was less
244 obvious within cores LOI13-3 and LOI13-4. Finally, the bottom of all the cores displayed a
245 faintly laminated unit labeled U2 in which dark lamina showed fluctuating MS values. The
246 interface between U1b and U2 across the core transect is very likely related to the reflector R1
247 considering the reflector depth (approximately 40 cm at the site of core LOI13-5) and the
248 geometry of the acoustic unit U1 that can be seen both in the axis of the river and in the
249 zigzag transect (Figure 2 and Figure 3).

250 A multi-proxy characterization of the sedimentary units identified within core LOI13-
251 2 is shown in Figure 3. Grain-size analyses revealed a sandy-silt sediment with Dx50 ranging
252 from a mean of 30 μm in U2 to a mean of 50 μm in U1a and U1b. At 27 cm depth, the U1-U2
253 interface (i.e., reflector R1) was also underlined by a slight rise in TOC content from 4 to 5%.
254 These parameters together supported the correlation across the millstream between the
255 acoustic unit U1 and the sedimentary units U1a and U1b. The dry bulk density displayed
256 values ranging from 0.7 to 1 $\text{g}\cdot\text{cm}^{-3}$ and was unrelated to the sedimentary units.

257



258

259 Figure 3: Multi-proxy sediment analyses along the core LOI13-2. MS, DBD, TOC and Dx50 were
 260 magnetic susceptibility, dry bulk density, total organic carbon and mean grain-size respectively. The
 261 spectral map represents the first-derivative of spectrophotometry measures (FDS, see Figure S2 for
 262 details). The chronology is indicated with a temporal scale on the right-hand side and is based on the
 263 ^{137}Cs downcore distributions. The two clear ^{137}Cs peaks are related to the Chernobyl accident dated
 264 from 1986 and to the maximum of nuclear atmospheric tests dated from 1963, respectively. Note that
 265 the sedimentation rate is almost constant throughout the core. In this dataset, error bars varied between
 266 4.9 and 6.3 Bq.kg^{-1} for the $^{210}\text{Pb}_{\text{ex}}$ and between 0.4 and 0.9 Bq.kg^{-1} for the ^{137}Cs .

267 4.3. Chronology

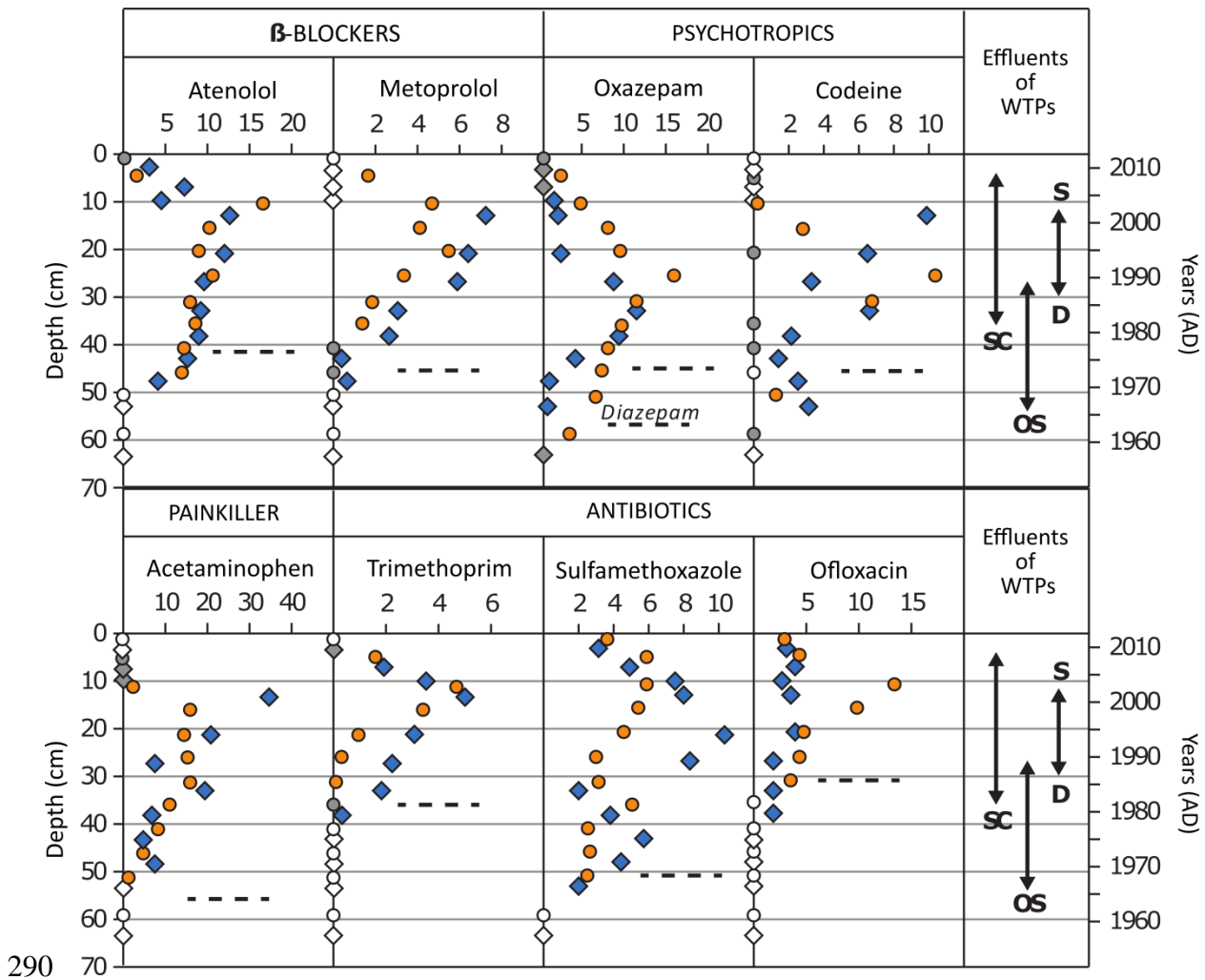
268 The down-core ^{137}Cs distribution revealed two clear peaks (Figure 3). The first peak,
 269 located in the 27-30 cm depth interval (31.9 Bq.kg^{-1}), can be related to the Chernobyl accident
 270 dated from 1986, whereas the second peak, measured in the 54-57 cm depth interval (27.6
 271 Bq.kg^{-1}), can be related to the atmospheric testing of nuclear weapons that took place during

272 the 1950s and 1960s, with a maximum activity in 1963 (UNSCEAR, 2000). The use of these
273 two chronomarkers suggests a constant accumulation rate of 1.1 cm.y^{-1} over the last 50 years.
274 Below 7 cm, the $^{210}\text{Pb}_{\text{ex}}$ activities decrease with increasing depth (Figure 3). Whereas the top
275 core may be affected by bioturbation, the exponential decay of the $^{210}\text{Pb}_{\text{ex}}$ activities between
276 7.5 and 37.5 cm indicated a sedimentation rate of 1.4 cm.y^{-1} , which is in relatively good
277 agreement with the sedimentation rate determined from ^{137}Cs concentrations.

278 **4.4. Pharmaceuticals**

279 The analysis of PPs along cores LOI13-2 and LOI13-6 firstly revealed that the
280 concentration range was the same whatever the PP investigated, varying from 1 to 30 ng.g^{-1} of
281 dried sediments (Figure 4). The vertical distribution of PPs is guided by a first occurrence, at
282 different depths for each PP, followed by an ongoing detection until the top of the core. In
283 general the lowest contamination levels were in the deepest layers. The highest concentration
284 was observed at depths varying between 35 and 10 cm followed by a progressive decrease in
285 U1a. Secondly, concentrations exhibited significant variations all along the core profiles. For
286 example, sulfamethoxazole was first quantifiable at 51 cm (i.e. $\text{AD}1968 \pm 1.6$) in core LOI13-
287 2 and then continuously detected until the top of the core with a maximum concentration at 11
288 cm (i.e. $\text{AD}2004 \pm 1.5$).

289



291 Figure 4: Distribution of PPs along core LOI13-2 (orange circles) and core LOI13-6 (blue diamonds)
 292 according to depth and years (age-depth model performed on core LOI13-2). Concentrations are
 293 expressed in $\text{ng}\cdot\text{g}^{-1}$ of dried sediment. Grey markers indicate concentrations below LOQ whereas
 294 empty markers indicate concentrations below LOD. The dashed line refers to the market authorization
 295 date (found on the ANSM website, www.ansm.fr) for each compound unless otherwise specified. The
 296 evolution of WTP effluents feeding the Dhuy River is indicated on the right-hand side of the figure
 297 (Figure 1 and Table 1).

298

299

300 5. Discussion

301 5.1. Mobility and persistence assessment

302 Mobility and persistence are both important parameters that must be taken into
 303 account when investigating the occurrence of PPs within the sedimentary archive. Mobility
 304 within sedimentary records can be assessed according to the organic carbon normalized
 305 partition coefficient values (K_{oc}) of each PP (Tamtam et al., 2011). Among the selected PPs,
 306 there is a wide chemical variety and hence different mobility. The selected PPs are classified
 307 between mobile to immobile (Table 2), which could potentially affect our reading of the
 308 results. According to literature data (Table 2), sulfamethoxazole is classified from mobile to
 309 moderately mobile whereas ofloxacin can be considered as the least mobile PP, due to its
 310 strong affinity with both organic and inorganic surfaces. Nevertheless, none of the selected
 311 PPs can be considered as highly mobile, and ofloxacin was used as a control in comparison to
 312 the other PPs.

313 Table 2: Relevant parameters assessing the mobility and persistence of PPs in sediments, with K_{ow} the
 314 octanol/water partition coefficient, K_d and K_{oc} respectively the solid/water partition and the normalized
 315 organic carbon partition both expressed in $L\ kg^{-1}$, mobility classification taken from ref.^h, the
 316 biodegradability based on the MITI-test (BIOWINTM, equation 6), and Removal the extreme removal
 317 (min-max) values of selected pharmaceuticals in conventional activated sludge treatment plants

318 ^a Barron et al. (2009)

319 ^b Al-Khazrajy and Boxall (2016)

320 ^c Schaffer et al. (2012)

321 ^d Stein et al. (2008)

322 ^e Drillia et al. (2005)

323 ^f Boxall et al. (2002)

324 ^g Löffler et al. (2005)

325 ^h Hollis (1991)

326 ⁱ Verlicchi et al., (2012)

327 ^j Wick et al., (2009)

Compound	Log K_{ow}	K_d	Log K_{oc}	Mobility class ^h	Biodegradability	Removal ⁱ capacity
Acetaminophen	0.46	32 ^a	2.9 ^a	slightly mobile	0.51	86.4-99.9
Atenolol	0.16 /1.37	15 ^a ; 9.3 ^b	2.3 ^b ; 2.6 ^a ; 3.1 ^c	moderately-slightly mobile	0.23	10-76

Codeine	1.2	14.1 ^d	2.5 ^d	moderately mobile	0.06	82
Metoprolol	1.79	20 ^a	2.75 ^a	slightly mobile	0.14	10-31
Ofloxacin	0.84	3554 ^e	4.6 ^f ; 4.7 ^e	immobile	0.0001	13-84
Oxazepam	2.2	23.5 ^d ; 2.2 ^g	2.2 ^g ; 2.7 ^d	moderately mobile	0.03	-10-35 ^l
Sulfamethoxazole	0.9	8 ^a ; 0.9 ^d ; 37.6 ^e	1.3 ^d ; 2.3 ^a ; 2.7 ^e	mobile-moderately mobile	0.006	10-100
Trimethoprim	0.4	26 ^a	2.9 ^a ; 3.6 ^c	slightly mobile	0.16	-46-53

328

329 In contrast to mobility, persistence is usually assessed by solar and biological
330 degradation impacts (Klaminder et al., 2015). However, the operating conditions used to
331 determine the behavior of organic pollutants with respect to degradation are too different for
332 these studies to be compared. We therefore estimated the degradability of each PP following
333 the output of the same model, the MITI-test (see supplementary data for details or Tunkel et
334 al., 2000), a component of the BIOWINTM model in the EPISuiteTM. According to these
335 outputs, the only biodegradable PP (i.e. Biodegradability > 0.5) among the ones selected is
336 acetaminophen (Blair et al., 2015; Lam et al., 2004). Moreover, the persistence of PPs should
337 be assessed with respect to their removal rate in WTPs. In Table 2, the removal values for
338 selected PPs concern WTPs that are similar to those present on the millstream (i.e.
339 conventional activated sludge treatment). The literature data are very variable, except for
340 acetaminophen, which has an extremely high removal value (i.e. mostly up to 99% in the
341 literature). This is in accordance with the output of the MITI-test, indicating that
342 acetaminophen is sensitive to degradation. However, due to its high consumption,
343 acetaminophen is frequently detected in European surface waters (Loos et al., 2009).
344 Concerning the other PPs investigated, the removal values are difficult to evaluate given the
345 strong variations in the available literature. Nevertheless, only an incomplete removal of these
346 PPs is achieved by WTPs.

347 5.2. Historical distribution

348 The chronology was determined on core LOI13-2. Thus, only the comparison between
349 PP concentrations and core chronology will be discussed hereafter. As noted in previous
350 studies, the market authorization date can be confronted with the first occurrence of each PP
351 dated in the sedimentary record (Tamtam et al., 2011; Klaminder et al., 2015). Figure 4 shows
352 that the market authorization date was systematically older than the date of the first
353 occurrence of a PP. Yet some exceptions can be noticed such as oxazepam, atenolol and
354 codeine. For example, as oxazepam is both a PP and a degradation product of diazepam
355 (Almeida et al., 2015), the degradation pattern of diazepam (sales authorized since 1953) may
356 play a significant role in the first occurrence of oxazepam within sedimentary archives.
357 Concerning atenolol, an occurrence before the market authorization date is unexpected due to
358 its moderate mobility. However, mobility remains the only consistent assumption to explain
359 occurrences older than the market authorization date (Tamtam et al., 2011). Finally,
360 concerning codeine, the path leading to its market authorization date is very hard to follow.
361 According to the ANSM website, the sale of codeine was authorized in 1973, but codeine is
362 also derived from opium. It is therefore not possible to assign a precise date to the arrival of
363 this molecule within aquatic environments.

364 The trend of ofloxacin, the least mobile PP (Table 2), revealed a first occurrence
365 around 1984, consistent with its market authorization date in 1980, followed by a progressive
366 increase until the early 2000s (Figure 4). The two most recent sample analyses (dating from
367 2007 and 2012, respectively) revealed a significant decrease in concentration. This trend
368 observed on ofloxacin also applied to trimethoprim, metoprolol, atenolol and acetaminophen.
369 However, oxazepam and codeine, which are two psychotropic drugs, exhibited a different
370 trend, with maximum concentration before 1989.

371 The good agreement between market authorization date and the first occurrence of the
372 majority of the PPs confirms that the sorption of PPs onto sediment preserves the PP from

373 degradation during decades, which is a well-known process in numerous environments (Keil
374 et al., 1994; Ingerslev and Halling-Sørensen, 2001) and confirmed in other studies in
375 analogous areas (Tamtam et al., 2011; Klaminder et al., 2015). Moreover, the oldest samples -
376 that is to say the deepest part of the concentration profiles - are also consistent with the market
377 authorization dates of the selected PPs, followed by a significant increase in the consumption
378 of PPs in France, particularly between the 1950s and the 1990s (Chauveau, 2002).
379 Nevertheless, the drop in concentrations observed in the upper part of the cores is more
380 surprising and could be due to two developments: (i) the enhancement of WTP removal
381 techniques, and (ii) the discharge of effluents in the rivers concerned, which has been
382 significantly reduced over the last twenty years (Table 1).

383 **5.3. Impact of WTP management**

384 Despite the presence of PPs in the karstic groundwater (Joigneaux, 2011), we assume
385 that effluents from WTPs in the Dhuy river catchment area are the main sources of PPs
386 because the karstic springs (Figure 1) do not feed the Loiret river substantially (Geo-hyd,
387 2009).

388 Among the two above-mentioned developments, the evolution of removal techniques
389 is difficult to assess whereas the history of effluent discharge in the selected rivers is well-
390 known. The millstream studied here was first concerned by a major event dated from 1989
391 with the deflection of the effluent discharges of the WTP closest to the sampled sediments
392 (i.e. site OS, Figure 1, Table 1). This event can be observed on the sedimentary profile, but
393 not for all the PPs. The deflection of the discharge of effluents from OS clearly affected the
394 trend of both codeine and oxazepam (Figure 4), the two psychotropic PPs. As this WTP also
395 contributes to the transport of suspended sediments in the river, it is possible to link this
396 deflection with the onset of U1 associated to a shift in sedimentological and geochemical
397 signals (Figure 3). Concerning the other PPs, the concentration trend is more affected by the

398 deflection of discharge from two smaller WTPs, SC and S-D in 2009 and 2003 respectively.
399 Overall, there is a strong impact of human management on the chemical quality of sediments
400 because of the non-correlation between the concentrations in sediment and the global increase
401 in the consumption of PPs. Another possibility is to characterize good markers for different
402 sources, since psychotropic drugs seem to be more sensitive to urban effluent deflection,
403 whereas the other PPs are more sensitive to rural WTP deflection.

404 **5.4. Spatial Distribution**

405 The distance between the two cores is approximately 1 km (Figure 2). This distance
406 precluded a direct application of the dating on LOI13-6. Nevertheless, several sedimentary
407 data (Figure 3), combined with the PP concentrations (Figure 4), give insight into the spatial
408 distribution of PPs since vertical patterns are found and repeated within the two cores. For
409 instance, an offset of sedimentary rate can be noticed in the lower part of the core for
410 oxazepam and sulfamethoxazole (Figure 4). Conversely, in the upper part of the two cores,
411 the concentrations are closer at the sampling interval used. The decrease in PP concentrations
412 since the 2000s is visible on the two records at the same depth. More generally, the
413 concentration range of each PP is the same between the two cores, suggesting that the
414 contamination is potentially widespread at the millstream scale.

415 **5.5. Assessment of the stock of pollution**

416 Despite the low concentrations and the decreasing trends observed for recent decades,
417 sediment contamination by PPs remains a potential threat for aquatic environments. In this
418 context, the quantification of PPs was combined with the sediment volume calculation
419 (Figures 2 and S1) to assess sedimentary contamination at the millstream scale. The volume
420 of U1, estimated at $21,000 \pm 100 \text{ m}^3$, was converted into a mass of $18,900 \pm 90 \text{ t}$, applying a
421 mean density of 0.9 g.cm^3 (Figure 3). This amount of sediment was then converted into a

422 mass of PPs on the basis of the mean concentrations in U1a and U1b covering the 1989 - 2013
423 period (Figure S2). Results indicate that during this period, a total of 763 ± 565 g of the
424 studied PPs accumulated in the sediments trapped in the millstream. These results likely
425 represent a small part of the global pharmaceutical pollution in this millstream. Nevertheless,
426 they constitute one of the first applications of sediment volume calculations coupled to
427 pharmaceutical pollution to assess the contamination at the reservoir scale, and although the
428 mass of PPs in this study represents a first approximation, it provides useful indicative
429 information for the future management and potential decommissioning of reservoirs where the
430 storage of contaminated sediments may be an issue for the quality of the aquatic environment.
431 Furthermore, assessment of the uppermost sedimentary unit (U1, Figure S1) indicates the
432 amount of contaminants discharged over the last 30 years. This assessment of PPs represents
433 the first step toward estimating the historical concentrations in river sediments, based on
434 solid/water partition of each PP. Moreover, the long-term consumption of each PP in the
435 catchment could be estimated based on the PPs concentrations in sediments, compared to the
436 existing approach of calculating catchment influent mass loads using only a limited temporal
437 period that is subject to significant daily variations (Thiebault et al., 2017).

438 **6. Conclusion**

439 This work provides new insights into the contamination of urban sediments by PPs.
440 For the first time, the occurrence of PPs from various therapeutic groups was assessed and
441 quantified following a temporal and spatial distribution. Pharmaceutical products are
442 differently recorded in the sediment from a chronological perspective depending on their
443 mode of consumption and their main origin. The chronological distribution of
444 pharmaceuticals was consistent with the global increase in drug consumption all around the
445 world and particularly in France since World War II. In the deepest part of the core (dated

446 between the 1950s and 1980s), PPs concentrations increased continually following their initial
447 market authorization date.

448 However, an opposite concentration trend was observed in recent decades, that can
449 first of all be assigned to the enhancement of WTP removal techniques (i.e. addition of
450 secondary treatment) but also to the management of the effluent discharge points during the
451 last 30 years. The nearest and largest WTP discharge points in the studied millstream were
452 deviated from the Dhuy river to the Loire river. As a result, the chemical quality of these
453 sediments regarding PP concentration was significantly enhanced in more recent years, thus
454 proving that the anthropogenic impact on river waters can be limited if appropriate
455 management is conducted. Yet, not all the PPs are sensitive in the same way to the deviation
456 of the discharge point of WTPs. Whereas the deflection of effluent discharges from the
457 catchment of the urban WTP impacted the concentrations of psychotropic PPs, the effluent
458 deflections of 3 more rural WTPs impacted antibiotic occurrences, indicating that the link
459 between consumption and concentrations in sediments is not instinctive and should be
460 carefully addressed depending on the investigated site. Finally, while the chemical quality of
461 the sampled river was enhanced, the diversion of the effluents probably increased the
462 contamination of the Loire river.

463 The quantification of PPs along the two cores gives similar concentrations, at least in
464 the same order of magnitude, making it possible to estimate a stock of less than 1 kg of
465 contaminants trapped within the sediments into the millstream. While a good ecological
466 continuity can be achieved for many rivers by removing small dams, such management
467 practices may damage the chemical status of superficial waters with the release of PP-
468 contaminated sediments. Finally, as the adsorption onto organic or inorganic surfaces seems
469 to preserve organic contaminants from degradation during several decades, the detection of
470 some PPs in recent sediments, such as antibiotics (i.e. poorly biodegradable and poorly

471 mobile), has potential for use as chronomarkers for age-depth modeling to assess
472 anthropogenic impacts on river and sediment quality.

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