

Article (refereed) - postprint

Hanamoto, Seiya; Nakada, Norihide; Juergens, Monika D.; Johnson, Andrew C.; Yamashita, Naoyuki; Tanaka, Hiroaki. 2018. **The different fate of antibiotics in the Thames River, UK, and the Katsura River, Japan.**

Environmental Science and Pollution Research, 25 (2). 1903-1913.

<https://doi.org/10.1007/s11356-017-0523-z>

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<https://doi.org/10.1007/s11356-017-0523-z>

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1 The different fate of antibiotics in the Thames River, 2 UK and the Katsura River, Japan

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13 **Abstract**

14 Little is known about the mechanisms influencing the differences in attenuation of antibiotics
15 between rivers. In this study, the natural attenuation of four antibiotics (azithromycin,
16 clarithromycin, sulfapyridine, and sulfamethoxazole) during transport along the Thames River, UK,
17 over a distance of 8.3 km, and the Katsura River, Japan, over a distance of 7.6 km were compared.
18 To assist interpretation of the field data, the individual degradation and sorption characteristics of
19 the antibiotics were estimated by laboratory experiments using surface water or sediment taken
20 from the same rivers. Azithromycin, clarithromycin, and sulfapyridine were attenuated by 92%,
21 48%, and 11% in the Thames River stretch. The first-order decay constants of azithromycin and
22 sulfapyridine were similar to those in the Katsura River, while that of clarithromycin was 4.4 times
23 higher. For sulfamethoxazole the attenuation was limited in both rivers. Loss of sulfapyridine was
24 attributed to both direct and indirect photolysis in the Thames River, but to only direct photolysis
25 in the Katsura River. Loss of azithromycin and clarithromycin was attributed to sorption to sediment
26 in both rivers. The probable explanation behind the difference in loss rates of clarithromycin

27 between the two rivers was considered to be sediment sorption capacity.

28 **Keywords**

29 antibiotics, natural attenuation, sorption, sediment, direct photolysis, indirect photolysis

30 **Introduction**

31 Antibiotics have been detected worldwide in various environmental media including fresh water
32 (Kolpin et al. 2002; Kasprzyk-Hordern et al. 2008; Shimizu et al. 2013), coastal water (Managaki
33 et al. 2007; Jia et al. 2011; Shimizu et al. 2013), and sediments (Feitosa-Felizzola et al. 2009; Blair
34 et al. 2013; Xu et al. 2014). Due to the potential risk to aquatic organisms (Cooper et al. 2008;
35 Boxall et al. 2012; Cizmas et al. 2015) and possible links to antibiotic resistance (Ågerstrand et al.
36 2015), antibiotics are considered some of the most important emerging contaminants in aquatic
37 environments. Studies on antibiotic resistant bacteria have shown their wide prevalence in natural
38 environments, including drinking water resources (Sharma et al. 2016), which is one of the most
39 important challenges to the health care sector in the 21st century (Carvalho and Santos 2016).
40 Therefore, in order to assess their risk and to aid in their management, the environmental fate and
41 behaviour of antibiotics should be modeled.

42 In the aquatic environment, antibiotics may be attenuated by physical, chemical, and/or
43 biological processes. Studies on the natural attenuation of antibiotics during river transport suggest
44 rapid removal is possible for some macrolide, quinolone, and tetracycline antibiotics (Hanamoto et
45 al. 2013; Barber et al. 2013; Luo et al. 2011). However, the reported attenuation of pharmaceuticals,
46 including antibiotics, often differs between rivers (Li et al. 2016; Radke et al. 2010; Kunkel et al.
47 2011; Dickenson et al. 2011; Acuña et al. 2015; Aymerich et al. 2016).

48 To understand natural attenuation, we must identify which are the key factors or processes that
49 can explain the different loss rates between rivers. The mechanisms influencing the different
50 attenuation of antibiotics between rivers have been estimated based on general characteristics of
51 rivers such as hydrological, meteorological, and water quality parameters (Li et al. 2016; Dickenson
52 et al. 2011; Acuña et al. 2015). Li et al. (2016) observed the attenuation of pharmaceuticals in four
53 European rivers, and suggested that shallow depth and low turbidity made the photochemical
54 attenuation more efficient in a small river, compared with larger rivers. However, in most cases, no
55 obvious explanation was found for the difference between rivers. It is presumed river characteristics,
56 such as the composition of sediments, dissolved matter, and microbial communities (which are
57 related to sorption, photolysis, and biodegradation processes, respectively), are determining the
58 different fates of antibiotics among rivers. But to date little research has been carried out to resolve
59 the importance of these processes on the different fates.

60 Thus, the aim of this study was to identify the mechanisms influencing the different fate and
61 behaviour of selected antibiotics between rivers. The natural attenuation of the antibiotics in the
62 Thames River (UK) were compared with the observations for the Katsura River (Japan) examined
63 previously (Hanamoto et al. 2013). To help distinguish the roles of the local degradation potential

64 and sorption characteristics for the antibiotics in both rivers, laboratory experiments and model
65 estimations were used. The antibiotics studied were two macrolides (azithromycin and
66 clarithromycin) and two sulfonamides (sulfapyridine and sulfamethoxazole). Their physical
67 properties are summarized in the Supporting Information (SI) Table S1. It is desirable to be in a
68 better position to predict the fate of pharmaceuticals and particularly antibiotics in rivers. In this
69 case the significance of the fate and behaviour of the same compounds in very different
70 rivers/climates/topologies was examined essentially to ask how predictable is their loss?

71 **Materials and methods**

72 **Site descriptions**

73 Samplings were conducted along an 8.3 km stretch of the Thames River (Fig. 1A), between site
74 1 (51°42'55"N, 1°14'11"W) and site 3 (51°40'14"N, 1°16'8"W), in Oxfordshire. The stretch
75 receives water from Littlemore Brook (site 2), where treated wastewater is discharged 1.6 km
76 upstream of site 2. The Katsura River stretch (7.6 km) receives water from two wastewater
77 treatment plants (sites a-c) and two tributaries (sites d and e) (Fig. 1B). There is little vegetation
78 and no significant additional inflows along the two river stretches. The Thames River stretch
79 catchment is mostly composed of limestone, clay/mudstone, and sandstone (Smith 2013), while the
80 Katsura River stretch catchment is mostly granite, chalk, clay/mudstone, and sandstone (Ministry
81 of Land, Infrastructure and Transport 2014). Most residents in both catchments are connected to
82 the respective sewer system. These stretches were selected because they are highly impacted by
83 treated wastewater, and antibiotics concentrations were expected to be higher there than elsewhere
84 in each river. The general characteristics of the Thames River stretch and the Katsura River stretch
85 are summarized in Table 1.

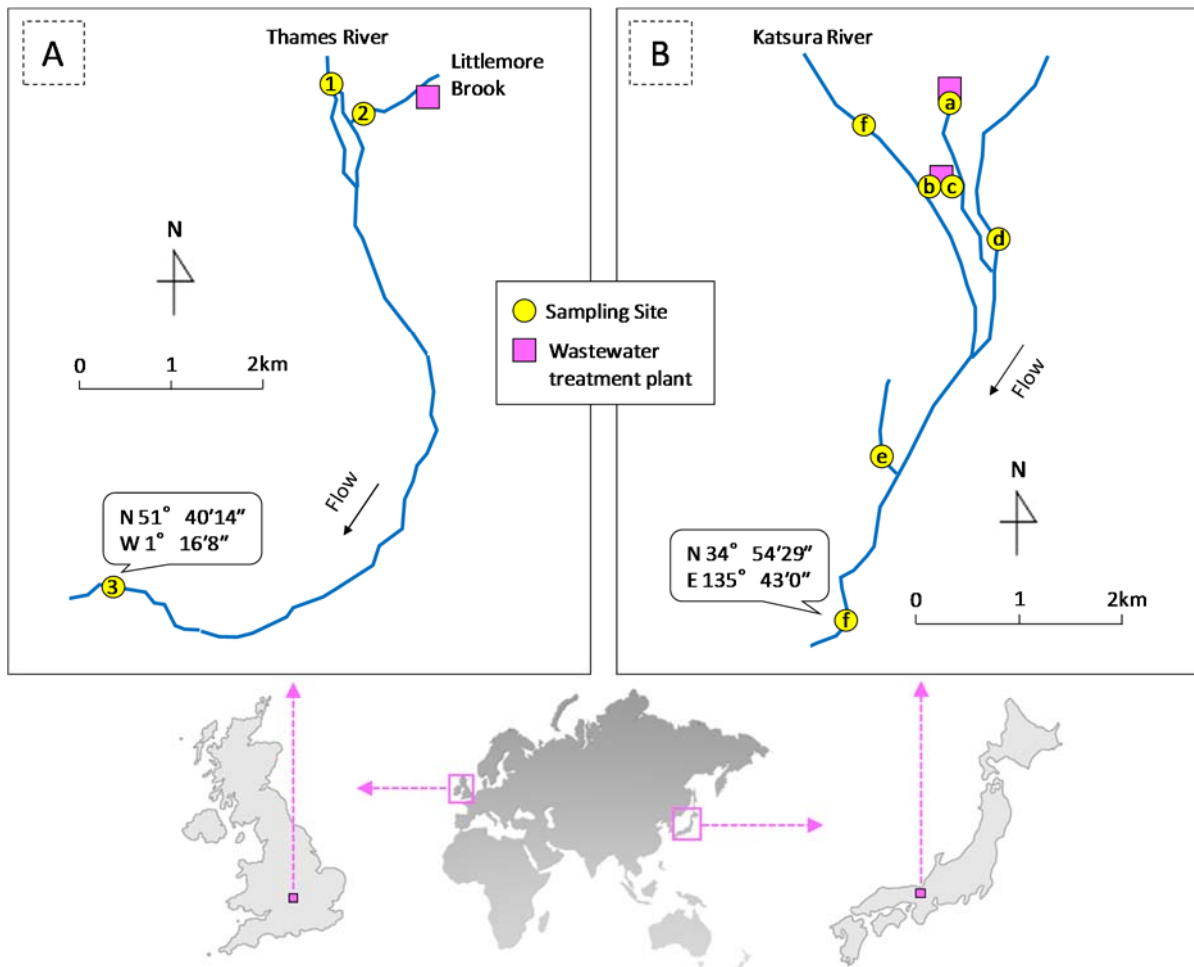
86

Table 1 General characteristics of the Thames and Katsura River stretches.

		Thames River stretch	Katsura River stretch
Water quality parameters ^a	temperature (°C) ^b	18.0 (16.0 - 19.9)	21.6 (9.4 - 27.3)
	pH ^b	8.0 (7.8 - 8.1)	7.4 (7.4 - 7.5)
	suspended solids (mg/L) ^b	8.9 (6.9 - 23.6)	5.8 (3.5 - 19.8)
Hydrological conditions ^a	flow rate (m ³ /s) ^c	7.6 (4.0 - 49.5)	22.4 (18.0 - 36.6)
	flow velocity (m/s) ^c	0.35 (0.26 - 0.90)	0.54 (0.50 - 0.67)
	depth (m) ^c	0.53 (0.39 - 1.38)	0.48 (0.44 - 0.56)
	hydraulic radius (m) ^c	0.52 (0.38 - 1.30)	0.47 (0.44 - 0.55)
	friction velocity at sediment-water interface (m/s) ^c	0.031 (0.024 - 0.067)	0.048 (0.045 - 0.058)
	travel time (h)	6.5 (2.6 - 8.9)	8.6 (7.0 - 9.4)
	Stretch length (km)	8.3	7.6

^aMedian (minimum - maximum) in the field studies (n = 7, the Thames River; n = 6, the Katsura River), obtained as described in SI "General Characteristics of the Rivers". ^bData at the most downstream site in each stretch. ^cAverage of several sites in each stretch.

88
89



90
91

Fig. 1 Locations of sampling sites on the (A) Thames River stretch and (B) Katsura River stretch.

92 **Field study**

93 The details of the sampling method in the Thames River stretch are described elsewhere (Nakada
94 et al. 2017). Surface water samples were collected at three sites, once or twice a summer between
95 2012 and 2015, yielding a total of seven samplings. All the samples were collected hourly for 24 h
96 by using automatic water samplers (ISCO Avalanche, ISCO 6712, Hach Sigma SD 900 or Bühler
97 Montec Xian 1000). For the samples collected in 2012 and 2013, the 24 hourly samples were
98 combined to 12 two-hourly samples and subjected to the antibiotics analysis separately. Since the
99 diurnal variations of mass loading of the selected antibiotics were not significant (see “Natural
100 attenuation of antibiotics ~” below), the samples collected in 2014 and 2015 were collected as 24
101 h composite samples by combining the 24 hourly samples. The samples were stored in plastic
102 bottles with ascorbic acid at 1.0 g/L in darkness (to reduce sample deterioration) and taken to the
103 laboratory. The samples were stored in a dark room at 4 °C until treatment. Details of the sample
104 analysis are described in SI “Analysis of Antibiotics”. Briefly, samples were filtered and
105 concentrated by solid-phase extraction within two days of collection and the four selected
106 antibiotics and the antiepileptic agent carbamazepine were measured by ultra-performance liquid
107 chromatography coupled to a tandem mass spectrometer and quantified by a surrogate method
108 (Narumiya et al. 2013). Carbamazepine can be considered as a relatively conservative tracer and so
109 was used to estimate the flow rate (see below).

110 The mass balance approach was used to estimate the attenuation of the antibiotics. The amount
111 of an antibiotic attenuated during the transport along the stretch relative to total mass loadings from
112 sites 1 and 2 is defined as the attenuation rate (equation 1). In addition, since most attenuation
113 processes in a river can be regarded as first-order reactions (see “Laboratory experiment and model
114 estimation” described below), a first-order decay constant was determined by equation 2, using the
115 Goal Seek function of Microsoft Excel. The flow rate at site 3 was estimated by subtracting the
116 reported flow rate at the Ock River from that at Sutton Courtenay on the Thames River (Centre for
117 Ecology and Hydrology). The flow rates at sites 1 and 2 were estimated by using the mass balance
118 of carbamazepine, which is persistent in aquatic environments (Nakada et al. 2008; Yamamoto et
119 al. 2009) (equations 3 and 4). The laboratory experiments indicated that carbamazepine was also
120 persistent in the Thames River stretch (see “Biotic and abiotic degradation ~” described below).
121 The travel time was calculated from length and flow velocity, which was estimated from the flow
122 rate using an empirical general relationship for rivers in the UK (Round et al. 1998).

$$123 \quad R_a = \frac{C_1 Q_1 + C_2 Q_2 - C_3 Q_3}{C_1 Q_1 + C_2 Q_2} \times 100 \quad (1)$$

$$124 \quad C_3 Q_3 = C_1 Q_1 e^{-k_a t_1} + C_2 Q_2 e^{-k_a t_2} \quad (2)$$

$$125 \quad Q_1 = \frac{Q_3 (C_{CBZ_3} - C_{CBZ_2})}{C_{CBZ_1} - C_{CBZ_2}} \quad (3)$$

$$126 \quad Q_2 = Q_3 - Q_1 \quad (4)$$

127 where R_a is attenuation rate (%); k_a is first-order decay constant (h^{-1}); C is concentration of an

128 antibiotic in surface water (ng/L); C_{CBZ} is concentration of carbamazepine in surface water (ng/L);
129 Q is flow rate (m^3/s); t is travel time to site 3 (h); and the subscripts are site IDs in Figure 1.

130 The sampling and calculation methods for the Katsura River stretch were similar to those for the
131 Thames River stretch. The samples were collected from seven sites (Fig. 1B) three times in summer
132 and three times in winter between 2011 and 2012. The details were described in our previous study
133 (Hanamoto et al. 2013).

134 **Laboratory experiment and model estimation**

135 Indirect photolysis, biodegradation, and sorption tests, and model estimations of direct photolysis
136 for the selected antibiotics and carbamazepine (test compounds) were carried out. Carbamazepine,
137 a compound used to estimate the flow rate in the field study, was included here to test its persistence
138 in the river stretches studied.

139 ***Direct photolysis estimation***

140 Direct photolysis represents degradation of a compound derived from direct absorption of light
141 by the compound, which follows a first-order reaction (Direct photolysis rate in water by sunlight
142 1998). Direct photolysis rate constants of the test compounds in the Thames River stretch under
143 average summer conditions were estimated by an equation proposed in our previous study
144 (Hanamoto et al. 2013). The equation considers the attenuation of sunlight in the atmosphere and
145 water, and was derived from equations proposed by Zepp et al. (1977) and Tixier et al. (2002) (see
146 SI equation S2). The parameters used in the estimation were set as follows. Reported values
147 (Hanamoto et al. 2013) were used for the photochemical properties (i.e., quantum yields and molar
148 absorption coefficients). The depth of water was estimated from the reported flow rate (Centre for
149 Ecology and Hydrology), estimated flow velocity (Round et al. 1998), and river width measured in
150 Google Maps[®], assuming the river cross-section to be a rectangle. To determine light penetration
151 in water, we collected surface water at site 3 under low flow conditions in 2013 summer and
152 measured the absorptivity between 290 and 490 nm with a UV-Vis spectrophotometer (UV-2500PC,
153 Shimadzu, Kyoto, Japan). The measured value was used for the light absorption coefficient of the
154 water body. Reported values at latitude 50°N in summer under clear sky (Direct photolysis rate in
155 water by sunlight 1998) were used for the spectrum of sunlight at the water surface. Since we could
156 not obtain any monitoring data for UVB or UVA in the UK, those measured in Kyoto city, Japan
157 (Project for monitoring sunlight intensity) were substituted to estimate the average fraction of
158 sunlight blocked by the clouds. Theoretical values for sky radiation were used for the fraction of
159 sunlight reflected at the water surface and the path length of sunlight in the water (Zepp et al. 1977).
160 Since there was little overhanging vegetation along the river stretch, the fraction of sunlight shaded
161 by plants was set to 0. Direct photolysis rate constants in the Katsura River stretch were estimated
162 using the same equation under average summer and winter conditions, because the field studies in
163 the stretch were conducted in both summer and winter (see Table 1). The parameters used in the
164 estimation were obtained in our previous study (Hanamoto et al. 2013) for the Katsura River stretch.
165 The parameters for the Thames and Katsura river stretches are summarized in SI Table S3.

166 **Indirect photolysis test**

167 Indirect photolysis represents degradation of a compound driven by reactive species (e.g., singlet
168 oxygen and hydroxyl radical) produced under light irradiance to dissolved matter (e.g. humic
169 substance and nitrate) in surface water. Indirect photolysis rate constants of test compounds in the
170 river stretches were assessed by applying U.S. Environmental Protection Agency's (USEPA)
171 harmonized test guideline 835.5270 (Indirect photolysis screening test 1998) to the surface water
172 with an assumption of extrapolation as described below. Grab samples collected at site 3 in the
173 Thames River stretch were brought to the laboratory, and filtered through a 1 µm pore size glass
174 fiber filter (GF/B, Whatman, UK) to prevent sorption of test compounds to suspended solids. A
175 phosphate buffer solution (10 mM) at pH 7.8 (same as the surface water) was prepared with
176 ultrapure water. The test compounds were added to the filtered surface water and phosphate buffer
177 to give an initial concentration of 5 µg/L each. The 10 mL solutions in 20 mL quartz tubes were
178 exposed to natural sunlight under a clear sky in the daytime for 4 h. The test compounds before and
179 after the exposure were analyzed as described above. The change in concentrations in darkness was
180 negligible (data not shown). The surface water sample was collected in 2015 summer under low
181 flow condition (7th percentile in 2010-2014, obtained in Centre for Ecology and Hydrology), and
182 the experiment was conducted in duplicate in an open space at the Centre for Ecology and
183 Hydrology, UK (51°36'9"N, 1°6'45"W), within two days of the sample collection. The same
184 laboratory conditions and procedures were applied to the surface water sample collected at the
185 Miyamae Bridge, the most downstream site in the Katsura River stretch (site g in Figure 1B), under
186 low flow condition (9th percentile in 2010-2014, obtained in Ministry of Land, Infrastructure and
187 Transport; website). The sunlight exposure to the Katsura River water was conducted in open space
188 at the Research Center for Environmental Quality Management, Japan (35°0'9"N, 135°53'24"E).

189 Since the indirect photolysis is generally a pseudo-first-order reaction (Indirect photolysis
190 screening test 1998), the indirect photolysis rate constant in the test tube was estimated by
191 subtracting the first-order rate constant in the phosphate buffer from that in the surface water
192 (equations 5 and 6). To extrapolate from the rate in the quartz test tube to that in the Thames River
193 stretch, the ratio of sunlight absorbed by influential dissolved matter (i.e., dissolved matter
194 producing reactive species influential on the antibiotics degradation) in the stretch to that in the
195 tube is needed. Calculated sunlight intensities in the tubes with surface water and phosphate buffer
196 were similar (data not shown) due to the short light path length in the tube. Most dissolved matter
197 and also antibiotics absorb sunlight mainly in the 300-400 nm range (Hanamoto et al. 2013).
198 Therefore, the ratio was estimated by dividing the direct photolysis rate constants of antibiotics
199 estimated in the stretch ($k_{d_{env}}$) by those observed in the test tube ($k_{pw_{tub}}$), assuming that the shape
200 of the solar action spectrum of the influential dissolved matter is similar to that of the antibiotics.
201 The indirect photolysis rate constant in the Thames River stretch was then estimated using this ratio
202 (equation 7).

203
$$k_{sw/pw_{tub}} = -\frac{1}{t_i} \ln \frac{C_{sw/pw_{aft}}}{C_{sw/pw_{bef}}} \quad (5)$$

204
$$k_{i_{tub}} = k_{sw_{tub}} - k_{pw_{tub}} \quad (6)$$

205
$$k_{i_{env}} = \frac{k_{i_{tub}} k_{d_{env}}}{k_{pw_{tub}}} \quad (7)$$

206 where k is first-order rate constant (h^{-1}); C is concentration in the water (ng/L); t_i is exposure
 207 time (h); k_i is indirect photolysis rate constant (h^{-1}); k_d is direct photolysis rate constant (h^{-1});
 208 and subscripts tub , env , sw , pw , bef and aft are values in the test tube, aquatic environment, surface
 209 water, ultrapure water, before the exposure, and after the exposure.

210 **Biodegradation test**

211 To assess the degradation of test compounds resulting from the activity of microorganisms which
 212 live in surface water in the river stretches, a simple simulation test was conducted with reference
 213 to USEPA's harmonized test guideline 835.3170 (Shake Flask Die-Away Test 1998). Surface water
 214 grab samples collected at site 3 in the Thames River stretch were brought to the laboratory. The test
 215 compounds were first dissolved in ultrapure water and then added to the river water samples to give
 216 an initial concentration of $0.5 \mu\text{g/L}$ of each. The solutions were incubated at $20 \pm 1 \text{ }^\circ\text{C}$ in the dark
 217 on a rotating shaker at 100 rpm for 24 h. The test compounds in the dissolved phase before and
 218 after the incubation were analyzed as described above, and those in the particulate phase were
 219 analyzed as described in SI "Analysis of Antibiotics". The amount of a compound in the solution
 220 (i.e., total of dissolved and particulate phase) that was lost in the incubation relative to the amount
 221 before the incubation was defined as a biodegradation loss (equation 8). Sorption of test compounds
 222 to glassware was negligible (Hanamoto et al. 2013). The change in suspended solids concentration
 223 during the incubation was negligible (data not shown). The surface water samples were collected
 224 twice in 2015 summer under low flow conditions (7th and 12th percentile of the 5 year daily flow
 225 data 2010-2014, obtained in Centre for Ecology and Hydrology), and the experiments were
 226 conducted in duplicate within a day after the sample collection. The same laboratory conditions
 227 and procedures were applied to the surface water samples collected at the Miyamae Bridge in the
 228 Katsura River stretch (site g in Figure 1B), although the experiment was conducted only once using
 229 sample collected under high flow condition (83rd percentile of the 5 year daily flow data 2010-2014,
 230 obtained in Ministry of Land, Infrastructure and Transport; website). Since the biodegradation of a
 231 compound of low concentration is a first-order reaction (Shake Flask Die-Away Test 1998), a
 232 biodegradation rate constant was determined by equation 9.

233
$$L_b = \frac{M_{bef} - M_{aft}}{M_{bef}} \times 100 \quad (8)$$

234
$$k_b = -\frac{1}{t_b} \ln \left(1 - \frac{L_b}{100} \right) \quad (9)$$

235 where L_b is biodegradation loss (%); k_b is biodegradation rate constant (h^{-1}); M is mass of a
 236 compound in the solution (i.e., total of dissolved and particulate phase) (ng); t_b is incubation time
 237 (h); and subscripts bef and aft are values before and after the incubation.

238 **Sorption test**

239 Since the sorption rate of a compound to river sediment is difficult to determine at laboratory
240 scale due to complications by hydrological factors such as hyporheic exchange, the sediment-water
241 partition coefficient was measured to evaluate sediment sorption capacity. The sediment-water
242 partition coefficients of test compounds were estimated in accordance with OECD test guideline
243 No. 106 (Adsorption–desorption using a batch equilibrium method 1995). Sediment grab samples
244 were collected from the top 5 cm in the Thames River stretch - two mixture sites and in the Katsura
245 River stretch - three mixture sites in 2014 summer under low flow conditions and brought to the
246 laboratory, air-dried, and passed through a 2 mm sieve. The solvent used for the sorption
247 experiments with the sediments was surface water from the Katsura River at the Miyamae Bridge
248 (site g in Figure 1B) which was filtered through a 1 µm pore size glass fiber filter, and to which
249 0.02% sodium azide was added to minimize microbial activity. Sediments (0.15-0.20 g) and river
250 water (50 ml) were put into glass centrifuge tubes and the test compounds were added to give an
251 initial concentration of 200 ng/L each. The tubes were then rotated at 20 ± 1 °C in the dark. After
252 two days, the test compounds in the water and sediment were analyzed as described above and in
253 SI “Analysis of Antibiotics” respectively. We had previously observed that the sediment-water
254 equilibrium for the test compounds was nearly reached within two days and that changes in
255 concentration without sediment were negligible (Hanamoto et al. 2013). The sediment-water
256 partition coefficients were determined by dividing the concentration in the sediment by that in the
257 water (equation 10). A higher coefficient indicates a greater sorption capacity of the sediment. The
258 experiment was conducted in duplicate.

$$259 \quad K_p = \frac{C_{seq}}{C_{w_{eq}}} \quad (10)$$

260 where K_p is sediment-water partitioning coefficient (L/kg); $C_{w_{eq}}$ is concentration in the water at
261 equilibrium (ng/L); and C_{seq} is concentration in the sediment at equilibrium (ng/kg).

262 **Results and discussion**

263 **Natural attenuation of antibiotics in the rivers Thames and Katsura**

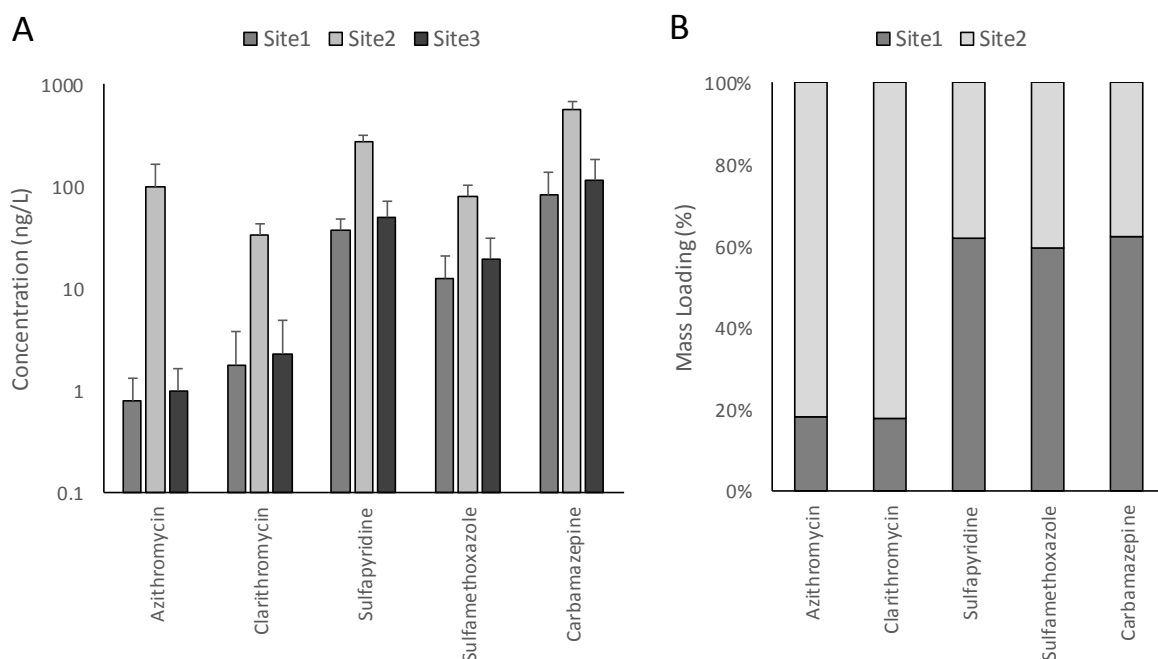
264 Concentrations of the selected antibiotics and carbamazepine in the Littlemore Brook (site 2), a
265 wastewater-impacted tributary, were much higher than those in the Thames River (sites 1 and 3,
266 Fig. 2A). The mass loadings from site 2 were higher than those from site 1 for azithromycin and
267 clarithromycin, and vice versa for the others (Fig. 2B). This difference between compounds would
268 be mainly attributable to the attenuation of the two macrolides during the transport in the stretch
269 upstream of site 1. Coefficients of variations (CVs) of concentrations within a day at the major
270 sources (i.e., site 2 for the two macrolides and site 1 for the others) were low (below 20%, see SI
271 Figure S1), indicating diurnal variation of concentration would not produce substantial errors in
272 estimating attenuations and flow rates by equations 1-4. Higher CVs of the two macrolides in sites

273 1 and 3 might be attributed to temporal variability of attenuation in the Thames River.

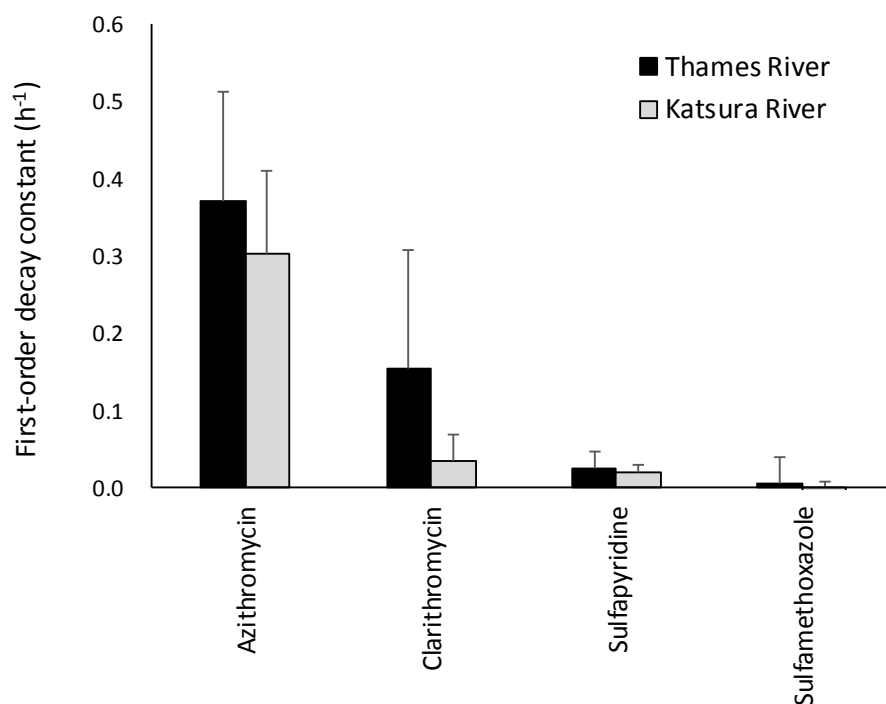
274 The attenuation rates for the Thames River stretch were low for the two sulfonamides,
275 sulfamethoxazole (-2% on average) and sulfapyridine (11%), and higher for the two macrolides,
276 clarithromycin (48%) and azithromycin (92%), indicating substantial losses for the macrolides. The
277 estimated average half-lives in the Thames River stretch were 29.2, 4.5, and 1.9 h for sulfapyridine,
278 clarithromycin, and azithromycin, respectively. The decay constants in the Thames River were 4.4,
279 1.2, and 1.2 times higher than those in the Katsura River for clarithromycin, azithromycin, and
280 sulfapyridine, respectively (Fig. 3). Thus, the fate of clarithromycin, in particular, was very
281 different between the rivers. Though there was no statistically significant difference in the decay
282 constants of antibiotics between the rivers even for clarithromycin ($p = 0.087$), this was attributable
283 to their high fluctuation between sampling days and the limited number of samples. The fluctuation
284 of the decay constants would be mainly driven by the daily variation in environmental factors such
285 as sunlight intensity and flow velocity (see “Mechanistic considerations for ~” described below).
286 Since surface water samples were collected as composite samples, the sampling error is minimal.

287 The limited attenuation of the sulfonamides in both rivers is similar to several observations in
288 Swedish rivers (Bendz et al. 2005; Li et al. 2016), American rivers (Barber et al. 2013), and a
289 Spanish river (Aymerich et al. 2016). No report on the decay constant was found for the investigated
290 macrolides, but that of erythromycin, a similar macrolide antibiotic, was reported in Iberian rivers
291 (0.17 h^{-1}) (Acuña et al. 2015). This is comparable to that of clarithromycin observed in the Thames
292 River stretch.

293



294
295 **Fig. 2** (A) Measured concentrations and (B) source distribution of antibiotics and carbamazepine
296 in the Thames River stretch. For A, the vertical bars denote means, and the error bars denote
297 standard deviations ($n = 7$). For B, the vertical bars denote medians ($n = 7$).



298
 299 **Fig. 3** Comparison of the first-order decay constants of the antibiotics between the Thames River
 300 stretch and the Katsura River stretch (means and standard deviations). The decay constant in the
 301 Katsura River was measured in a previous study (Hanamoto et al. 2013). For sulfapyridine, there
 302 was a significant difference between the summer and winter sampling campaigns in the Katsura
 303 River stretch (see SI Figure S2), so only the three decay constants obtained in summer were
 304 included, while all six decay constants were included for the other antibiotics. The number of
 305 samples was $n = 7$ for all antibiotics in the Thames River; $n = 3$ for sulfapyridine in the Katsura
 306 River; $n = 6$ for the other antibiotics in the Katsura River.

307
 308 **Biotic and abiotic degradation and sorption characteristics of antibiotics in the Thames and**
 309 **Katsura Rivers**

310 The direct photolysis rate constant of sulfapyridine was 0.011 h^{-1} in the Thames River stretch
 311 and 0.022 h^{-1} in the Katsura River stretch under average summer conditions and below 0.010 h^{-1} in
 312 the Katsura River stretch in winter. The lower rate constant in the Thames River compared to the
 313 Katsura in summer was mainly due to a higher light absorption coefficient of the water body (see
 314 SI Figure S3). The direct photolysis of sulfapyridine was attributed to desulfonation and/or
 315 denitrification, as well as hydroxylation of photo-oxidized heterocyclic rings (Baena-Nogueras et
 316 al. 2017). The rate constants of the other antibiotics were below 0.010 h^{-1} in both rivers.

317 The first-order rate constant of sulfapyridine observed in the indirect photolysis experiment in
 318 the Thames River water was 2.4 times higher than that in the ultrapure water, while the constant of
 319 sulfapyridine in the Katsura River water was similar to that in the ultrapure water (see SI Figure
 320 S4). The estimated indirect photolysis rate constant of sulfapyridine in the Thames River stretch
 321 was 0.015 h^{-1} , while that in the Katsura River stretch was below 0.010 h^{-1} . Previously, significant

322 indirect photolysis of sulfapyridine was reported in water from a constructed wetland, and this was
323 attributed not to nitrate but to a portion of dissolved organic matter (Challis et al. 2013). Since
324 there is no other study on indirect photolysis of sulfapyridine under sunlight, further mechanistic
325 studies should be conducted to elucidate constituents of dissolved organic matter determining the
326 indirect photolysis of sulfapyridine. The decrease in concentrations of the other antibiotics during
327 the sunlit experiment were below 20%, yielding indirect photolysis rate constants of below 0.010
328 h⁻¹ in both rivers.

329 The biodegradation losses derived from the laboratory experiments over 24 h were below 20%
330 and the biodegradation rate constants were below 0.010 h⁻¹ for all antibiotics in both rivers. The
331 reported biodegradation rate constants of the antibiotics, which were observed in Katsura River
332 water under the five-day incubation test, were also below 0.010 h⁻¹ (Hanamoto et al. 2013).

333 The relative sediment-water partitioning coefficients were azithromycin > clarithromycin >>
334 sulfapyridine > sulfamethoxazole in both rivers. The partitioning coefficients for the Thames River
335 sediments were 1.4 and 5.5 times (on average) higher than those for the Katsura River sediment for
336 azithromycin and clarithromycin, respectively. Since the two macrolides mostly exist in cationic
337 forms in surface water (Sibley et al. 2008), their sorption is likely to be due to coulombic attraction
338 to negatively charged surface sites on sediments (e.g. permanent negative charge on aluminosilicate
339 clays, deprotonated surface hydroxyl groups on sediment metal oxides, and deprotonated surface
340 hydroxyl or carboxylic acid groups on sediment organic matter, Vasudevan et al. 2009). Because of
341 the multiple sorption sites, sorption capacities of macrolides did not correlate with general
342 properties of soils (e.g. organic carbon content, cation exchange capacity) (Kodešová et al. 2015;
343 Srinivasan et al. 2014), and there is no related mechanistic study with sediments. Given the geologic
344 differences between the two catchments (see “Site descriptions” above), mineralogical
345 compositions of sediments seem to be different between the rivers, and this might have caused the
346 much higher partitioning coefficient of clarithromycin in the Thames River than in the Katsura
347 River. Therefore, further mechanistic studies should be conducted to elucidate sediment
348 constituents determining the sorption capacity of clarithromycin. As for the difference in
349 partitioning coefficients between the compounds, previous studies on sediments indicated that the
350 octanol-water partitioning coefficient (K_{ow}), the indicator of hydrophobicity of a compound, could
351 not explain the different sorptivity between pharmaceuticals (Yamamoto et al. 2009; Schaffer et al.
352 2012). Given the hydrophilic and ionizable properties of pharmaceuticals and negatively charged
353 surface sites on sediments, the fraction of pharmaceuticals existing in cationic form would be the
354 most influential factor differentiating antibiotics sorptivities. Therefore, the observed low
355 sorptivities of the two sulfonamides could be because they mostly exist in anionic or neutral forms
356 within the common environmental pH range (Gao and Pedersen, 2005).

357 For carbamazepine, all the degradation rate constants were below 0.010 h⁻¹, the partitioning
358 coefficient was not measurable due to its low sorptivity, and its predicted volatilization is expected
359 to be negligible (Hanamoto et al. 2013) in both rivers, indicating the validity of using it as a
360 conservative tracer to estimate the flow rate in the river stretches.

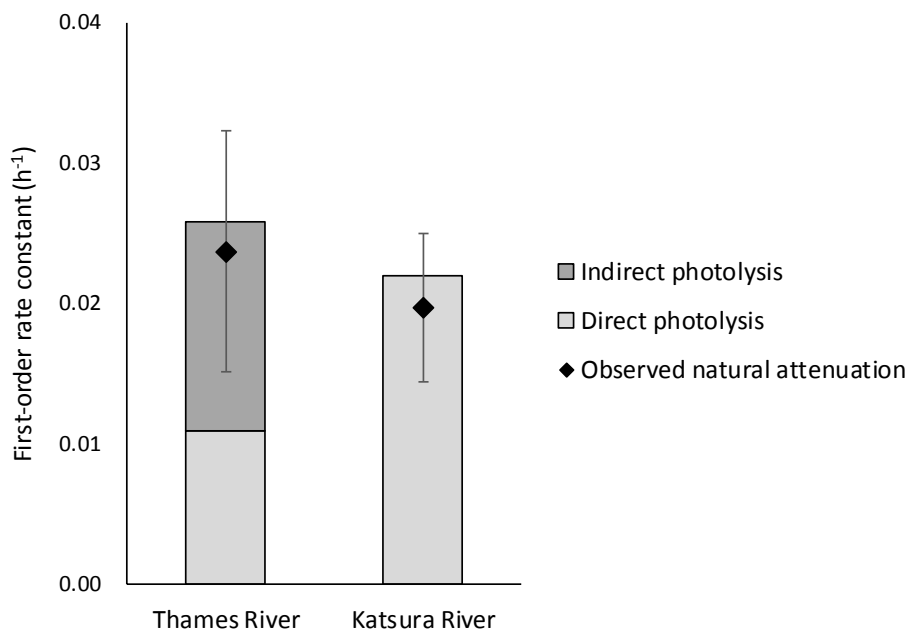
361 **Mechanistic considerations for the difference in attenuation between the rivers**

362 The degradation and volatilization (Hanamoto et al. 2013) rates were negligible and sediment
363 sorption capacity was low in both rivers for sulfamethoxazole, which is consistent with its limited
364 natural attenuation (see Figure 3). Since biodegradation and volatilization (Hanamoto et al. 2013)
365 were negligible and sediment sorption capacity was low in both rivers for sulfapyridine, its major
366 loss mechanism during transport along the river stretches was direct and/or indirect photolysis. The
367 decay constants of sulfapyridine observed in the river stretches were comparable to the sum of its
368 direct and indirect photolysis rate constants under average summer conditions (Fig. 4), indicating
369 the assumption made in extrapolating the indirect photolysis rate constants observed in the test
370 tubes into those in the river stretches (see “Indirect photolysis test” described above) did not
371 produce a substantial error in the estimate. Therefore, the attenuation of sulfapyridine can be
372 considered to be mainly due to both direct and indirect photolysis in the Thames River, but to only
373 direct photolysis in the Katsura River. The difference in indirect photolysis between the rivers is
374 attributable to constituents of dissolved organic matter (see “Biotic and abiotic degradation ~”
375 above).

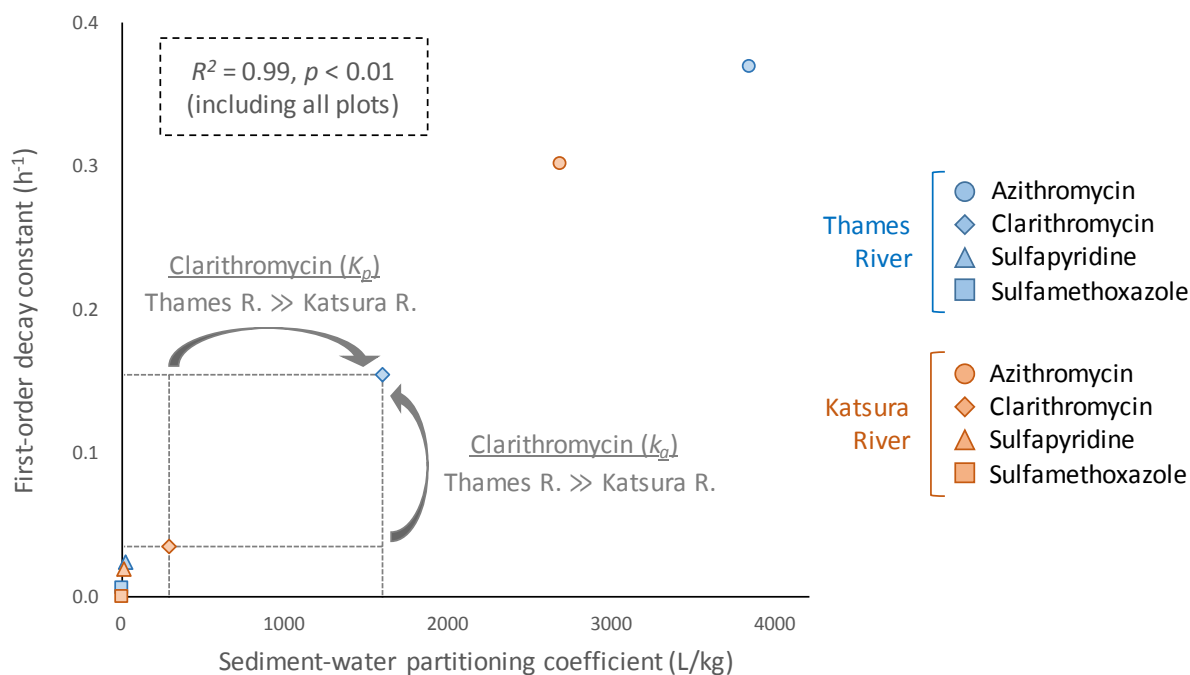
376 The degradation and volatilization (Hanamoto et al. 2013) rates were also negligible but the
377 sediment sorption capacity was high in both rivers for azithromycin and clarithromycin. Therefore,
378 the major loss mechanism during transport along the river stretches for the two macrolides would
379 be sorption to sediment, which is consistent with their reported high concentrations in sediments
380 (Blair et al. 2013; Feitosa-Felizzola et al. 2009; Xu et al. 2014; Luo et al. 2011). The much higher
381 decay constant of clarithromycin in the Thames River than in the Katsura River was attributable to
382 the higher sediment sorption capacity in the former, given their positive relationship shown in
383 Figure 5. Sorption rate constant of a compound (i.e., mass transferred from overlaying water into
384 sediment per unit time) is determined by a driving force, which is defined as difference in sediment
385 pore water and overlaying water concentrations, and a mass transfer coefficient (Thibodeaux 1996).
386 The sediment sorption capacity is an influential factor determining the driving force of sorption to
387 sediment.

388 Another possible factor differentiating the driving force of sorption between the rivers could be
389 biodegradation of clarithromycin within the sediment. Though there is no related study on
390 sediments, degradation rates of the macrolides were quite low within soils (Kodešová et al. 2016)
391 and biosolids-amended soils (Walters et al. 2010) (half lives were mostly over 100d), which
392 indicates biodegradation within sediment is likely to be low for clarithromycin. The sorption rate
393 constant is also affected by hydrological parameters such as flow velocity, hydraulic radius, and
394 hyporheic exchange, which are determinants of the mass transfer coefficient, and replacement of
395 surface sediment during high flow events, which affects the driving force. However, these
396 parameters could not explain the fact that the fate of only clarithromycin was very different between
397 the two rivers, since such parameters do not have substance-specific effects (Thibodeaux 1996).
398 The sorption to suspended solids was also not considered to be playing an important role given that
399 the concentration was low in both rivers (see Table 1). Thus, though there are some unquantified

400 factors, the sediment sorption capacity would be a key factor explaining the different fate of
401 clarithromycin between the rivers. Further studies to estimate the sorption rate constant should be
402 conducted and compared with the observed attenuation to help confirm this.
403



404 **Fig. 4** Comparison of the observed first-order decay constant and the estimated photolysis rate
405 constants for sulfapyridine under average summer conditions. Since there was a significant
406 difference in the decay constants between the summer and winter sampling campaigns in the
407 Katsura River stretch, the comparison was conducted under summer conditions. Means and standard
408 errors of measurements in summer sampling campaigns were substituted for the decay constants (n
409 = 7 for the Thames River; n = 3 for the Katsura River). The decay constant in the Katsura River
410 was measured in a previous study (Hanamoto et al. 2013).
411
412



413
 414 **Fig. 5** Comparison of the first-order decay constants (k_a) and the sediment-water partitioning
 415 coefficients (K_p) for antibiotics. Means were substituted for the decay constant ($n = 7$ for all
 416 antibiotics in the Thames River; $n = 3$ for sulfapyridine in the Katsura River; $n = 6$ for the other
 417 antibiotics in the Katsura River). The decay constants in the Katsura River were measured in a
 418 previous study (Hanamoto et al. 2013). A higher partitioning coefficient indicates a greater sorption
 419 capacity of the sediment. Since the partitioning coefficient of sulfamethoxazole was not available
 420 due to its low sorptivity, it was plotted as 0.
 421

422 Conclusions

423 The field study revealed that the decay constants in the Thames River were 4.4, 1.2, and 1.2
 424 times higher than those in the Katsura River for clarithromycin, azithromycin, and sulfapyridine
 425 respectively, while the attenuation was limited in both rivers for sulfamethoxazole. River
 426 characterization highlighted sediment sorption capacity played an important role in the different
 427 loss rates of clarithromycin between the two rivers. Attenuation of azithromycin was also attributed
 428 to sorption to sediment in both rivers. Both direct and indirect photolysis affected attenuation of
 429 sulfapyridine in the Thames River, while indirect photolysis was negligible in the Katsura River.
 430 These findings provide a better understanding of the key factors differentiating natural attenuation
 431 of antibiotics between rivers. Future work should focus on the sediment properties which determine
 432 the sorption capacity of the macrolides. In addition, more information is needed on the types of
 433 dissolved organic matter determining indirect photolysis of sulfapyridine, as well as on sorption
 434 rate constants of the macrolides in the rivers.

435 **Acknowledgments**

436 We thank the Japanese Ministry of the Environment and DEFRA-supported UK/Japan
437 cooperation. We thank Dr. SangJung Lee, Dr. Vimal Kumar, Dr. Masaru Ihara, and several
438 students of the Research Center for Environmental Quality Management, Kyoto University, for
439 their assistance in the laboratory and in the fieldwork. We thank Dr. Michael J. Bowes of the
440 Centre for Ecology and Hydrology for providing water quality data. This work was supported by
441 the Japan Society for the Promotion of Science (JSPS) Grants-in-Aid for Scientific Research (A)
442 (25257304, 23254003, and 26257302) and funding from the UK Natural Environment Research
443 Council.

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1 *SUPPORTING INFORMATION*

2 **The different fate of antibiotics in the Thames River, UK and the**
3 **Katsura River, Japan**

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16

17 Selected Antibiotics

18 Table S1. Physical properties of the selected antibiotics.

	therapeutic class	formula	molecular weight	pK_a	$\log K_{ow}$	Henry's Law constant ($\text{atm}\cdot\text{m}^3/\text{mole}$)
azithromycin	macrolide antibiotics	$\text{C}_{38}\text{H}_{72}\text{N}_2\text{O}_{12}$	749.0	8.7	4.0	5.3×10^{-29}
clarithromycin	macrolide antibiotics	$\text{C}_{38}\text{H}_{69}\text{NO}_{13}$	748.0	9.0	3.2	1.7×10^{-29}
sulfapyridine	sulfonamides	$\text{C}_{11}\text{H}_{11}\text{N}_3\text{O}_2\text{S}$	249.3	2.7, 8.4	0.35	1.1×10^{-13}
sulfamethoxazole	sulfonamides	$\text{C}_{10}\text{H}_{11}\text{N}_3\text{O}_3\text{S}$	253.3	1.6, 5.7	0.89	6.4×10^{-13}

19 There data were obtained from Syracuse Research Corporation, PhysProp Database (<http://www.srcinc.com/what-we-do/databaseforms.aspx?id=386>) or EPI Suite (<http://www.epa.gov/opptintr/exposure/pubs/episuite.htm>)

20

21 Analysis of Antibiotics

22 The four selected antibiotics and carbamazepine (test compounds) in the dissolved phase were analyzed
23 as follows. A 500 mL surface water sample was filtered through a glass fiber filter (GF/B, 1.0 μm , Whatman,
24 UK), and spiked with a surrogate standard mixture (containing 50 μg each of five isotopically labelled
25 pharmaceuticals) (Narumiya et al., 2013). The test compounds in the dissolved phase were concentrated by
26 solid-phase extraction through Oasis HLB cartridges (500 mg, 6cc, Waters, Japan). The cartridges containing
27 the sample concentrate and surrogate standards were stored in a refrigerator for up to a few weeks, then
28 extracted with 6 mL of methanol before being measured by ACQUITY ultra-performance liquid
29 chromatography system (Waters, USA) coupled to a Micromass Quattro micro API Tandem Quadrupole
30 system (Waters, USA). An ACQUITY BEH C18 column (1.7 μm , 2.1 mm \times 100 mm, Waters, USA) was
31 used for separation. Measured concentrations were quantified by the surrogate method (Narumiya et al.,
32 2013). Details of the analytical method and quantification limit of test compounds can be found in previous
33 studies (Okuda et al. 2009; Narumiya et al. 2013).

34 The test compounds in the particulate phase of the water and sediments were analyzed as follows.
35 Suspended solids trapped on a 1- μm pore size glass fiber filter or 0.20-g subsamples of the sediment were
36 spiked with a surrogate mixture, and accelerated solvent extraction (ASE) was performed with an ASE 200
37 system (Thermo Fisher Scientific Inc., Waltham, MA, USA) in 11-mL ASE cells under the following
38 conditions: 100 $^\circ\text{C}$, 10 min static, 3 cycles, 2000 psi, and 60% flush. Methanol/water (1:1, v/v) with 0.5%
39 (v/v) aqueous ammonia (pH = 11) was used as a solvent. The extracts were filtered through a cellulose filter
40 (ASE extraction filters, Thermo Fisher Scientific) installed at the bottom of the cell, diluted with ultrapure
41 water to reduce the concentration of methanol to <5% (v/v), and had ascorbic acid (2 g/L) and
42 ethylenediaminetetraacetic acid disodium (1 g/L) added to them. Solid-phase extraction was used to clean-
43 up the extracts by using Oasis HLB cartridge (200 mg, 6cc, Waters, USA). Test compounds were treated,
44 measured, and quantified as described above.

45 To test the analytical reproducibility of the test compounds in the dissolved phase, the following

46 experiment was conducted. Three ultrapure water samples (200 mL) were spiked with a surrogate standard
47 mixture containing 50 ng of each and the test compounds were analyzed as described above. The relative
48 standard deviations of absolute recoveries of the test compounds in the water samples ranged from 1.1%
49 (carbamazepine) to 12.8% (sulfapyridine), indicating that the analytical procedure was accurate. In addition,
50 to validate the method for sediment samples, the following experiment was conducted. The sediment sample
51 collected in the Katsura River was passed through a 2-mm sieve, and ten 0.20 g subsamples were obtained.
52 Five were spiked with a test compounds mixture containing 500 ng of each compound, and all ten were
53 additionally spiked with a surrogate mixture containing 50 ng of each. Then, the test compounds were
54 analyzed as described above. The accuracy of the analytical method was evaluated using the absolute and
55 relative recoveries (over extraction, solid-phase extraction, and measurement) of the spiked test compounds.
56 The absolute recoveries of the test compounds in the sediment ranged from 70.2% (carbamazepine) to 88.6%
57 (sulfamethoxazole); their relative recoveries ranged from 83.4% (carbamazepine) to 101.4% (azithromycin);
58 and the relative standard deviations of the relative recoveries ranged from 2.5% (carbamazepine) to 8.3%
59 (sulfapyridine). Thus, the analytical procedure of the sediment samples was also deemed accurate for the test
60 compounds.

61

62 **General Characteristics of the Rivers**

63 We measured water quality parameters at site 3 in the Thames River during the sampling in 2015, while
64 for the years 2012-2014 those measured within ± 2 days of our sampling 22.4 km downstream of site 3 in
65 Wallingford (Personal communication from Dr. Michael J. Bowes) were used. Since the inflows between site
66 3 and Wallingford are small compared with the flow at site 3, the water quality was similar (Table S2).
67 Reported data at the Miyamae Bridge (Ministry of land, infrastructure and transport) were used for the water
68 quality parameters in the Katsura River, though concentration of suspended solids was estimated from
69 turbidity (estimation accuracy, $R^2=0.91$).

70 The flow rates at site 3 in the Thames River were estimated by subtracting the reported flow rate at the
71 Ock River from those at Sutton Courtenay on the Thames River (Centre for Ecology and Hydrology). For
72 the Katsura River, the reported flow rates at the Miyamae Bridge were used (Ministry of land, infrastructure
73 and transport). The flow velocities in the Thames River were estimated using a general flow rate-flow
74 velocity relationship for rivers in the UK (Round et al. 1998), while those in the Katsura River were estimated
75 using the relationship developed in our previous study (Hanamoto et al. 2013). The depth of water in the
76 Thames River was estimated from the flow rate, the flow velocity, and river width measured in Google Maps[®],
77 assuming the river cross-section to be a rectangle, while that in the Katsura River was estimated using the
78 flow rate-water depth relationship developed in our previous study (Hanamoto et al. 2013). The hydraulic
79 radius was estimated from the river depth and width assuming the river cross-section to be rectangle for the
80 Thames River, while that in the Katsura River was estimated using the river cross-section diagram obtained
81 in our previous study (Hanamoto et al. 2013). Friction velocity at the sediment-water interface was estimated
82 by the equation S1, assigning a value of 0.025 to Manning's roughness coefficient. The travel time was
83 calculated from length and the flow velocity.

84

$$v_* = \frac{vn\sqrt{g}}{R^{1/6}} \quad (S1)$$

85 where v_* is friction velocity at the sediment-water interface (m/s); v is flow velocity (m/s); n is
 86 Manning's roughness coefficient (s/m^{1/3}); g is gravity acceleration (m/s²); and R is hydraulic radius (m).

87

88 Table S2. Comparison of water quality between Wallingford and site 3.

	date	time	temperature (°C)	pH	suspended solids (mg/L)
Wallingford ^a	August 24, 2015 ^b	14:40 ^b	18.4 ^b	7.9 ^b	10.8 ^b
Site 3	August 24, 2015	11:00	19.4	8.0	8.12

^a 22.4 km downstream of site 3. ^b Reference 2.

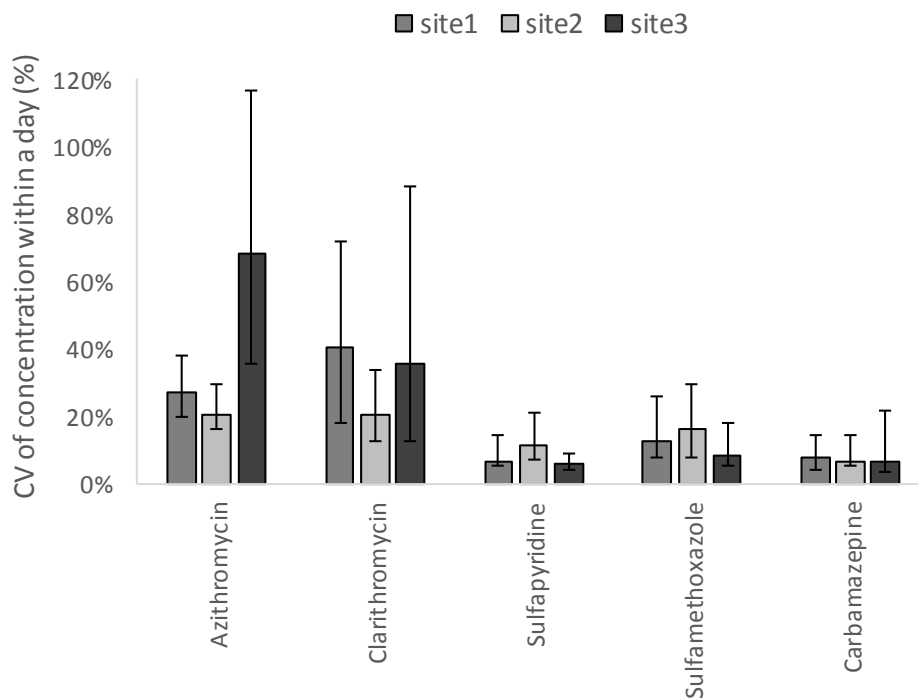
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91 Field Study

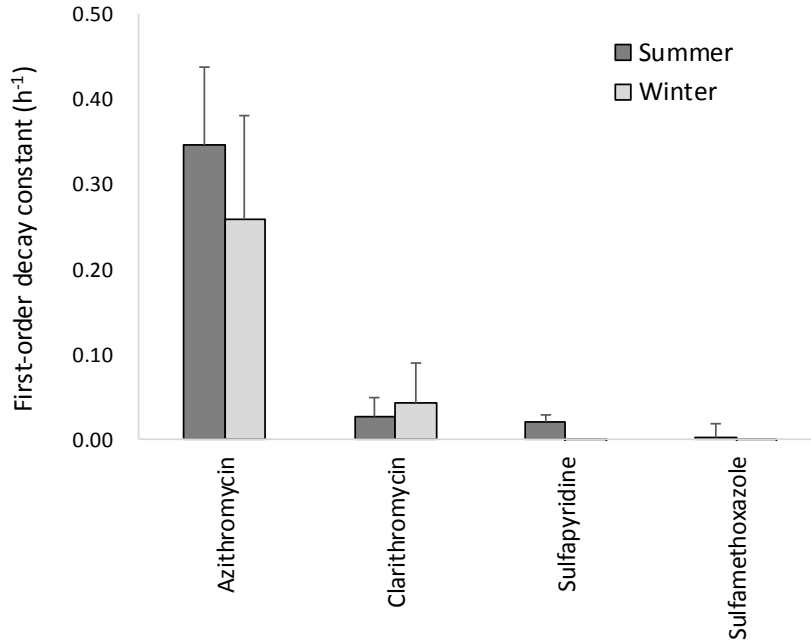
92 The diurnal variation in concentrations of the test compounds in the Thames River stretch are shown in
 93 Figures S1. Comparison of the first-order decay constants of the antibiotics in the Katsura River stretch
 94 obtained in summer and winter in a previous study (Hanamoto et al. 2013) are shown in Figure S2.

95



96

97 Figure S1. Coefficients of variation (CV) of concentration of antibiotics and carbamazepine within 24 hours
 98 (12 samples). The vertical bars denote median and the error bars denote maximum and minimum (n = 4).



99

100 Figure S2. Comparison of the first-order decay constants of the antibiotics in the Katsura River stretch
 101 obtained in summer and winter sampling campaigns in a previous study (Hanamoto et al. 2013). The
 102 vertical bars denote means and the error bars denote the standard deviation (n = 3 for both summer and
 103 winter).

104

105 Laboratory Experiments

106 The direct photolysis rate constant (k_p) of a compound in river i was estimated as

$$\begin{aligned}
 107 \quad k_{p_i} = \varphi \times & \left\{ \frac{UVB_i \times (1 - R_{UVB_i}) \times (1 - B_{UVB_i})}{UVB_t} \times \sum_{\lambda = 297.5}^{315} \frac{L_\lambda \times (1 - 10^{-\alpha_{\lambda_i} \times l_i}) \times \varepsilon_\lambda}{\alpha_{\lambda_i} \times D_i} \right. \\
 108 \quad & \left. + \frac{UVA_i \times (1 - R_{UVA_i}) \times (1 - B_{UVA_i})}{UVA_t} \times \sum_{\lambda = 315}^{490} \frac{L_\lambda \times (1 - 10^{-\alpha_{\lambda_i} \times l_i}) \times \varepsilon_\lambda}{\alpha_{\lambda_i} \times D_i} \right\} \quad (S2)
 \end{aligned}$$

109 where φ is quantum yield of the compound (-); UVB and UVA are sunlight intensity at Earth's surface in
 110 those wavelengths (W/m^2); R_{UVB} and R_{UVA} are fraction of sunlight reflected at the surface of the water
 111 body in those wavelengths (-); B_{UVB} and B_{UVA} are fraction of sunlight shaded by aquatic and overhanging
 112 vegetation in those wavelengths (-); UVB_t and UVA_t are theoretical sunlight intensity at Earth's surface
 113 in those wavelengths (W/m^2); L_λ is theoretical sunlight intensity at Earth's surface at wavelength λ (10^{-3}
 114 Einstein $cm^{-2} h^{-1}$); α_λ is light absorption coefficient of the water body at wavelength λ (m^{-1}); l is path
 115 length of sunlight in the water body (m); ε_λ is molar absorption coefficient of the compound at wavelength
 116 λ ($M^{-1} cm^{-1}$); and D is depth of the water (m). This equation was proposed in a previous study (Hanamoto
 117 et al. 2013).

118 The parameters used for the estimation of the direct photolysis constant in the Thames and Katsura river
 119 stretches are summarized in Table S3.

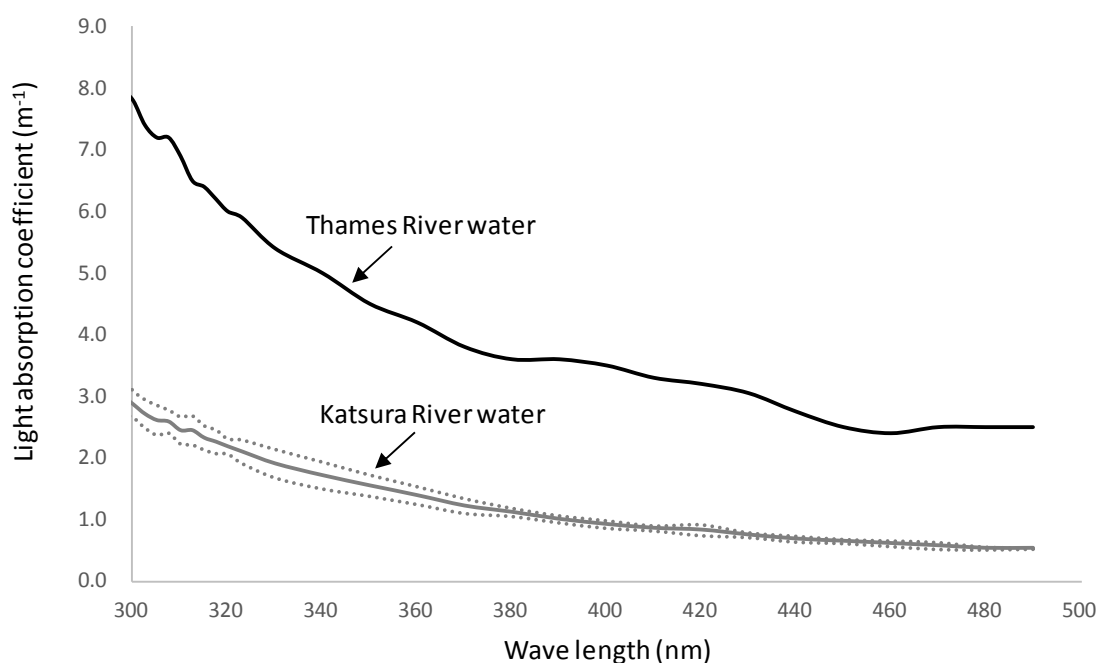
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121 Table S3. Parameters used for the estimation of the direct photolysis constants of test compounds in the
 122 Thames and Katsura river stretches.

		Outline of used data	
		Thames River stretch ^a	Katsura River stretch ^b
UVB, UVA	Sunlight intensity at Earth's surface in UVB or UVA	The ratios UVB/UVB_t and UVA/UVA_t were estimate from	Reported value in Kyoto city
UVB_t, UVA_t	Theoretical sunlight intensity at Earth's surface in UVB or UVA	observed and teoritical sunlight intensity in Kyoto city	Theoretical value at latitude 40°N
L_λ	Theoretical sunlight intensity at Earth's surface at wavelength λ	Theoretical value at latitude 50°N	Theoretical value at latitude 40°N
B_{UVB}, B_{UVA}	Fraction of sunlight shaded by plants in UVB or UVA	Since there is little overhanging vegetation along the river stretches, these were set to 0	
R_{UVB}, R_{UVA}	Fraction of sunlight reflected at water surface in UVB or UVA	Theoretical value for sky radiation	Theoretical value for sky radiation
l	Path length of sunlight in water body	Theoretical value for sky radiation	Theoretical value for sky radiation
α_λ	Light absorption coefficient of water body at wavelength λ	Measurements at site 3	Measurements at site f
D	Depth of water	Estimate from reported flow rate	Estimate from reported flow rate
ϕ	Quantum yield of the compound	Reported value	Reported value
ϵ_λ	Molar absorption coefficient of the compound at wavelength λ	Reported value	Reported value

^aDetails and references for the Thames River stretch were shown in the main text "Direct photolysis estimation". ^bDetails and references for the Katsura River stretch were shown in a previous study (Hanamoto et al. 2013).

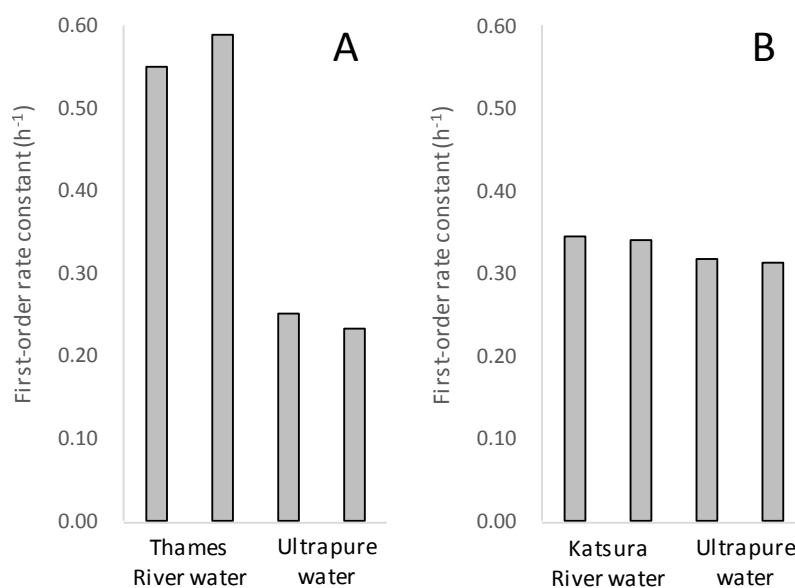
123
 124
 125 The light absorption coefficient of the surface water at site 3 in the Thames River was compared with those
 126 reported at the Miyamae Bridge in the Katsura River (Hanamoto et al. 2013) (Figure S3). The light absorption
 127 coefficient was much higher in the Thames River, which is attributable to higher suspended solids and
 128 dissolved organic matter in the Thames River.



130 Figure S3. The light absorption coefficient of surface water at site 3 in the Thames River compared with
131 those reported at the Miyamae Bridge in the Katsura River (Hanamoto et al. 2013). The solid line denote
132 means and dashed line denote standard deviations in year-round sampling for the Katsura River (n = 6),
133 whereas the absorption of the Thames River surface water was only measured once during low flow
134 conditions in 2013 summer.

135

136 The first-order rate constants of sulfapyridine observed in the indirect photolysis experiments are shown
137 in Figure S4.



138

139 Figure S4. The first-order rate constants of sulfapyridine observed in the indirect photolysis experiments
140 conducted in (A) the UK and (B) Japan. The experiment was conducted in duplicate. Irradiation times in the
141 UK and Japan were 4.0 h and 2.2 h, respectively. The concentrations decreases were 49% (ultrapure water
142 in Japan) to 90% (Thames River water). The concentration decreases of the other antibiotics and
143 carbamazepine during the experiment were not appreciable (< 20%) in all samples.

144

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