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1 **Title**

2 Benefits of ozonation before activated carbon adsorption for the removal of organic
3 micropollutants from wastewater effluents.

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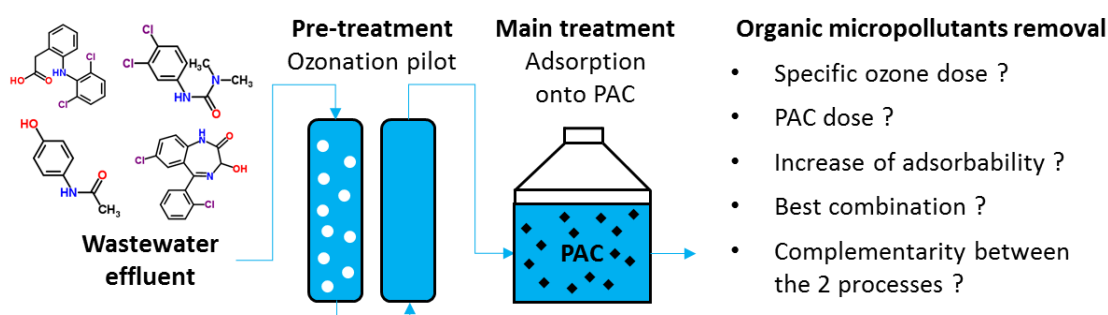
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21 Highlights

- 22 • OMPs adsorption increased after ozonation at high specific ozone doses only
- 23 • Ozonation performances were impacted by the presence of TSS
- 24 • Low specific ozone doses allowed the removal of OMPs refractory to adsorption
- 25 • Non-reactive OMPs toward ozone were well adsorbed onto PAC
- 26 • The best combination was 0.20-0.30 gO₃/gDOC and 10 mg/L PAC

27 Graphical abstract



28

29 Abstract

30 Advanced processes for the removal of organic micropollutants (OMPs) from wastewater
31 effluents include adsorption onto activated carbon, ozonation, or a combination of both processes.
32 The removal of 28 OMPs present in a real wastewater effluent was studied by ozonation coupled
33 to activated carbon adsorption and compared to a sole adsorption. The influence of the specific
34 ozone dose (0.09-1.29 gO₃/gDOC) and the influence of the powdered activated carbon (PAC)
35 dose (2, 5 and 10 mg/L) were first studied separately. OMPs removal increased with both the
36 specific ozone dose (up to 80% for a dose higher than 0.60 gO₃/gDOC) and the PAC dose.
37 Ozonation performances decreased in presence of suspended solids, which were converted to
38 dissolved organic carbon. A correction of the specific ozone dose according to the suspended
39 solids levels, in addition to nitrite, should be considered. The influence of ozonation (0.09, 0.22,

40 0.94 and 1.29 gO₃/gDOC) on OMPs adsorption was then assessed. OMPs adsorption didn't
41 change at low specific ozone doses but increased at higher specific ozone doses due to a decrease
42 in DOM adsorption and competition with OMPs. At low ozone doses followed by adsorption (0.22
43 gO₃/gDOC and 10 mg/L PAC), the two processes appeared complementary as OMPs with a low
44 reactivity toward ozone were well absorbed onto PAC while most OMPs refractory to adsorption
45 were well eliminated by ozone. Improved removals were obtained for all compounds with these
46 selected doses, reaching more than 80% removal for most OMPs while limiting the formation of
47 bromate ion.

48 **Keywords**

49 Wastewater advanced treatment; Organic micropollutants; Ozonation; Activated carbon
50 adsorption.

51 **1. Introduction**

52 Activated carbon adsorption is widely used for drinking water production and is transferred to
53 wastewater advanced treatment to remove organic micropollutants (OMPs) and particularly
54 pharmaceuticals compounds that are refractory to biological treatments before reuse or discharge
55 in aquatic ecosystems (Altmann et al., 2015b; Guillosoy et al., 2019; Meinel et al., 2015; Ruel et
56 al., 2011). Diverse types of activated carbon with different granulometries can be used, either
57 powdered activated carbon (PAC) (Altmann et al., 2015a; Kårelid et al., 2017; Meinel et al., 2016),
58 micro-grain activated carbon (μ GAC) (Alves et al., 2018; Mailler et al., 2016a) or granular
59 activated carbon (GAC) (Benstoem et al., 2017; Grover et al., 2011; Zietzschmann et al., 2016).
60 The use of μ GAC in fluidized bed was previously demonstrated to reach similar removals of
61 OMPs to PAC (~80-90%) (Mailler et al., 2015, 2016a) with a simpler and easier operation. However,
62 some pharmaceuticals such as diclofenac, erythromycin or sulfamethoxazole were poorly
63 removed (< 50%) by fluidized-bed μ GAC, due to specific physico-chemical properties decreasing

64 their adsorption (e.g. size, charge, hydrophobicity). The presence of dissolved organic matter
65 (DOM) in wastewater effluents also limits OMPs removal through competition for adsorption
66 sites and pore blockage (Newcombe et al., 2002; Zietzschmann et al., 2014).

67 Ozonation is currently the most used oxidation process for OMPs removal in drinking water
68 production and is also recently transferred to the advanced treatment of wastewater effluents (de
69 Wilt et al., 2018; Hollender et al., 2009; Margot et al., 2013; Zimmermann et al., 2011). Bourgin
70 et al. (2018) obtained removals higher than 80 % for the 12 OMPs used as indicator substances in
71 the Swiss legislation (Office fédéral de l'environnement, Division Eaux, 2014) over a whole
72 wastewater treatment plant upgraded with an ozonation step (0.55 gO₃/gDOC). Ozone is a
73 powerful oxidant which reacts rapidly with electron-rich moieties such as alkenes, amines and
74 aromatic rings (von Gunten, 2003a). The self-decomposition of ozone forms hydroxyl radicals
75 (\bullet OH) which can react with non-activated aromatic compounds, amides and alkanes. DOM and
76 OMPs are usually not mineralized but rather transformed into smaller compounds called ozonation
77 transformation products which are most of the time more biodegradable and less toxic (Hübner et
78 al., 2015; Völker et al., 2019). Oxidation by-products such as bromate (BrO₃⁻) and N-
79 nitrosodimethylamine (NDMA), which are suspected to be human carcinogens, can however be
80 formed depending on the ozone dose applied and on the presence of precursors. BrO₃⁻ can be
81 produced from the reaction of ozone with bromide ion and NDMA from specific precursor
82 compounds (e.g. anthropogenic organic compounds containing dimethylamine groups) (SgROI et
83 al., 2016; von Gunten, 2003b). An additional treatment step after ozonation such as sand filtration
84 or activated carbon filters is therefore required to reduce transformation products and oxidation
85 by-products.

86 In combination with activated carbon adsorption, ozonation appears as a complementary process
87 to remove compounds that are refractory to adsorption, as well as to oxidize DOM in order to
88 reduce the competition with OMPs onto activated carbon and then to limit the decrease of

89 activated carbon adsorption capacity over time. Indeed, Muller et al. (1996) reported that the
90 adsorption of the herbicide atrazine increased after pre-ozonation of DOM in river water. Matsui
91 et al. (1999) studied the effect of ozonation on the adsorption of DOM from peat water and
92 biological wastewater effluent and found a decreased in DOM adsorbability due to the formation
93 of weakly adsorbing compounds after ozonation. Zietzschmann et al. (2015) observed a decrease
94 in DOM adsorbability after the ozonation of a wastewater effluent due to a reduction of DOM
95 aromaticity, molecular size and hydrophobicity. This decrease led to a reduced adsorption
96 competition with OMPs and as a consequence improve the adsorption of OMPs on activated
97 carbon over time.

98 The coupling of ozonation and activated carbon is well known and already in use at full scale in
99 some wastewater treatment plants (WWTP) where ozonation is the main step for OMPs removal
100 and adsorption (mostly GAC) is an additional step for post-treatment. However, a coupled process
101 consisting of ozonation as a pre-treatment step (i.e. at lower doses than commonly employed)
102 followed by activated carbon adsorption as the main treatment step has never been considered.
103 The coupling could be an interesting option to increase the removal of OMPs refractory to
104 adsorption and for disinfection purposes while limiting the production of oxidation by-products,
105 but knowledge of its efficiency for real wastewater effluents, which are composed of a more
106 complex matrix than surface water, is limited. Moreover, available studies only investigated a
107 small number of OMPs spiked in wastewater effluents at relatively high concentrations (> 10
108 $\mu\text{g/L}$) (Zietzschmann et al., 2015). The aim of this study was to evaluate the removal of 28 OMPs
109 present in wastewater effluents by ozonation coupled to adsorption onto activated carbon as the
110 main treatment step, and to estimate the benefits provided by the coupling in comparison to
111 activated carbon adsorption alone. The influence of various specific ozone doses on OMPs
112 removal was first studied. The effect of the activated carbon dose was then evaluated with a
113 preliminary ozonated effluent. DOM evolution during ozonation and adsorption were examined

114 in parallel through UV₂₅₄ and fluorescence excitation/emission analyses. Different combinations
115 of specific ozone doses and activated carbon doses were tested to assess the potential beneficial
116 effect of ozonation on OMPs adsorption through DOM oxidation, to evaluate the complementarity
117 between both processes toward OMPs that are refractory to ozonation or adsorption and to find
118 the most promising combination of specific ozone and PAC doses.

119 **2. Material and methods**

120 *2.1. Ozonation pilot*

121 The ozonation experiments were conducted on a pilot comprising a cylindrical stainless-steel
122 transfer column (height = 100 cm, diameter = 25 cm, volume = 56 L) and a contact column of
123 similar dimensions (Figure S1). The ozone gas was produced in a corona-discharge tube from an
124 oxygen-enriched gas stream (90% O₂) produced by an oxygen concentrator (OnyxUltra, CAIRE
125 Inc.). The gas was sent to the transfer column from a porous diffuser at the bottom at counter-
126 current flow in the transfer column. The concentration of the ozone inlet and outlet gas of the
127 transfer column was measured by an ozone gas analyser (BMT 964, BMT Messtechnik GMBH).
128 The ozone gas flow was maintained at a constant value (2 NL/min) and the concentration of ozone
129 ranged from 8 to 45 g/Nm³. The ozonated water was sampled after the contact column for each
130 specific ozone dose in 20-L cleaned glass bottles. The water flow was maintained constant (10
131 L/min) and measured by a flowmeter. The hydraulic retention time in each column was about 5
132 min. The water pH (7.0-7.4) and temperature (20-25°C) were monitored by probes
133 (Endress+Hauser, Germany).

134 The pilot was fed continuously by nitrified water directly pumped from the second biofiltration
135 stage from the Seine-Centre WWTP (Colombes, France). The plant is operated by the Paris public
136 sanitation service (SIAAP) and treats wastewater for 800 000 inhabitants with a flow of 240 000
137 m³/day. The treatment is composed of a pre-treatment step (screening, grit and oil removal)

138 followed by the primary treatment (physico-chemical lamellar settling for the removal of
139 suspended particles, colloids and phosphorus) and the biological treatment (3 stages of
140 biofiltration for the removal of carbon and nitrogen). After treatment, the effluent is discharged in
141 the Seine River. The ozone dose transferred in the liquid phase was normalized by the dissolved
142 organic carbon concentration (DOC) after subtraction of the ozone consumption by nitrite ion
143 (NO_2^-) (Naumov et al., 2010) to obtain the value of the parameter “specific ozone doses” used all
144 along the article.

$$145 \text{ Specific } O_3 \text{ dose} = \frac{\text{Transferred } O_3 \text{ dose} - 1.04 * [\text{NO}_2^-]_{\text{consumption}}}{[\text{DOC}]}$$

146 With the specific O_3 dose in gO_3/gDOC , the transferred O_3 dose in $\text{gO}_3/\text{L}_{\text{wastewater}}$, $[\text{NO}_2^-]$
147 consumption in gNO_2^-/L and $[\text{DOC}]$ before ozonation in g/L .

148 An instantaneous ozone demand (IOD) of the nitrified water in nominal conditions was evaluated
149 at $3.8 \pm 0.5 \text{ mgO}_3/\text{L}$ ($\text{DOC} = 6 \text{ mgC/L}$, $\text{TSS} < 2 \text{ mg/L}$, $\text{NO}_2^- < 0.06 \text{ mg/L}$) following the method
150 developed by Roustan et al. (1998). The IOD was satisfied for specific ozone dose up to 0.7
151 gO_3/gDOC .

152 Apart from the water quality, the operating conditions did not change during the experiments
153 except the concentration of the ozone gas sent to the transfer column. An equilibration time of 15
154 min was allowed between the application of each specific ozone dose.

155 2.2. Experimental plan

156 The removal performances of 28 OMPs (19 pharmaceuticals, 6 pesticides, 2 hormones and 1
157 perfluorosulfonic acid – Table 1) were studied. A set of 5 experiments was conducted at various
158 specific ozone doses (ranging from 0.09 to 1.29 gO_3/gDOC) (Table 2). Nitrified water from the
159 Seine Centre WWTP was sampled on 2018-05-02, 2018-05-23, 2018-07-06, 2019-04-09 and
160 2019-08-07 (experiments 1 to 5, respectively). The quality of the wastewater effluent is generally

161 similar over this period of the year (Rocher et al., 2012). Experiment 1 and 2 were conducted to
162 study the influence of the specific ozone dose. Experiment 3, 4 and 5 were performed to study the
163 influence of the activated carbon dose, the performances of the coupled processes and the
164 influence of ozonation on adsorption performances, respectively. For experiments involving PAC
165 adsorption (experiments 3, 4 and 5), a μ GAC (CycleCarb 305, Chemviron Carbon) was crushed
166 and sieved to keep only particles between 50 and 63 μ m to guarantee a fast adsorption. This
167 activated carbon was chosen due to its previous use in a fluidized-bed industrial pilot (CarboPlus[®],
168 Saur) (Mailler et al., 2016a) and the chemical, physical and textural properties of this activated
169 carbon (Table S1) were determined in a previous study (Guillossou et al., 2020). An average
170 OMPs removal of 70% in this pilot was equivalent to 30% in batch with a similar PAC dose (10
171 mg/L) (Mailler et al., 2016b, 2015). Each desired PAC dose (2, 5 or 10 mg/L) was added to 10 L
172 of nitrified or ozonated water in 20 L flasks then adsorption tests were performed under fast
173 stirring for 30 min using a stainless-steel blades stirrer. The range of PAC doses was chosen from
174 a previous work which reported that the removal gain obtained when increasing the PAC dose
175 from 5 mg/L to 10 mg/L was more important than when increasing from 10 mg/L to 20 mg/L
176 (Mailler et al., 2016b). Prior to adsorption in experiment 5, the 28 studied OMPs were spiked at a
177 concentration of 100 ng/L to ensure their presence in the effluent after ozonation (high specific
178 ozone doses).

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184 Table 1. List of the 28 analysed OMPs, their reaction rate constant with ozone (k_{O_3}), molecular weight (M_w), hydrophobicity (log D) and charge at 20-25°C and pH
 185 = 7.0, and their limits of quantification (LOQ). The M_w , log D and charge were calculated using MarvinSketch (ChemAxon).

Compound	Use	k_{O_3} (L.mol ⁻¹ .s ⁻¹)	Reference for k_{O_3}	M_w (g/mol)	Log D	Charge	LOQ (ng/L)
Acetaminophen	Analgesic	2.6x10 ⁶	Hamdi El Najjar et al., 2014	151.2	0.91	0	0.5
Acetamiprid	Insecticide	0.25	Cruz-Alcalde et al., 2017	222.7	1.11	0	1
Atenolol	Betablocker	2.1x10 ³	Mathon et al., 2017	266.3	-2.24	+1	1
Carbamazepine	Anticonvulsive	3.0x10 ⁵	Huber et al., 2003	236.3	2.77	0	1
Ciprofloxacin	Antibiotic	1.9x10 ⁴	Dodd et al., 2006	331.3	-1.47	0	35
Clothianidin	Insecticide	-	-	249.7	-4.53	0	3
Diclofenac	Analgesic	1.0x10 ⁶	Huber et al., 2003	296.2	1.26	-1	0.5
Diuron	Pesticide	1.5x10 ¹	Mathon et al., 2017	233.1	2.53	0	1
Erythromycin	Antibiotic	7.0x10 ⁴	Mathon et al., 2017	734.0	1.20	+1	4
Estradiol	Hormone	3.0.x10 ⁶	Huber et al., 2003	272.4	3.75	0	7
Estrone	Hormone	1.5x10 ⁵	Deborde et al., 2005	270.4	4.31	0	2.5
Ibuprofen	Analgesic	9.6	Huber et al., 2003	206.8	1.69	-1	6
Imidacloprid	Insecticide	-	-	255.7	-1.95	0	2
Ketoprofen	Analgesic	0.4	Real et al., 2009	254.3	0.49	-1	1
Lorazepam	Anxiolytic	-	-	321.2	3.53	0	1.5
Naproxen	Analgesic	2.0x10 ⁵	Huber et al., 2005	230.3	0.18	-1	2
Norfloxacin	Antibiotic	4.5x10 ⁵	Ling et al., 2018	319.3	-1.57	0	5
Ofloxacin	Antibiotic	2.0x10 ⁶	Márquez et al., 2013	361.4	-0.05	-1	1
Oxazepam	Anxiolytic	1.0	Lee et al., 2014	286.7	2.92	0	2
PFOS	Additive	1.0	Arias Espana et al., 2015	500.1	-4.89	-1	0.5
Propranolol	Betablocker	1.0x10 ⁵	Mathon et al., 2017	259.3	-0.09	+1	1.5
Roxithromycin	Antibiotic	7.0x10 ⁴	Huber et al., 2003	837.1	0.92	+1	0.5
Sulfadiazine	Antibiotic	1.0x10 ⁵	Garoma et al., 2010	250.3	0.08	-1	1
Sulfamethoxazole	Antibiotic	2.5x10 ⁶	Huber et al., 2003	253.3	-0.11	-1	1
Tetracycline	Antibiotic	1.9x10 ⁶	Dodd et al., 2006	444.4	-0.57	0	22
Thiaclopride	Insecticide	-	-	254.7	2.06	0	2.5
Thiamethoxam	Insecticide	-	-	291.7	1.29	0	0.5
Trimethoprim	Antibiotic	3.0x10 ⁵	Dodd et al., 2006	290.3	0.89	+1	2

187 *2.3. Analytical procedures*

188 For each experiment, the concentrations of the 28 studied OMPs were analysed within 48 h
189 (Institut des Sciences Analytiques – Villeurbanne, France) by liquid chromatography coupled to
190 tandem mass spectroscopy. These micropollutants were chosen because of their frequency of
191 detection in the effluents of wastewater treatment plants in the Paris conurbation (Mailler et al.,
192 2015, 2014) or because they were molecules of recent interest (e.g. the 5 insecticides). Prior to
193 analysis, all samples were homogenized and filtered on 0.7 µm GF/F glass filters (Whatman).
194 Details of the analytical methods are described in supplementary information (Text S1) (Vulliet
195 et al., 2011). OMPs removal was calculated following the approach of Choubert et al. (2017),
196 which stated that the uncertainty of OMPs concentrations in wastewater generally ranges between
197 30% and 100% when the measured value is lower than 2.5 to 10 times the limit of quantification
198 (LOQ) of the compound. Therefore, when a compound was quantified at levels above 5 times the
199 LOQ in the nitrified water and above the LOQ after treatment, the removal was conventionally
200 calculated. When a compound concentration was above 5 times the LOQ in the nitrified water but
201 below the LOQ after treatment, the removal was estimated using LOQ/2. Removals were not
202 calculated when concentrations in the nitrified water and after treatment ranged between the LOQ
203 and 5 times the LOQ.

204 Conventional wastewater quality parameters were analyzed by the certified SIAAP laboratory:
205 total organic carbon (TOC), dissolved organic carbon (DOC), NH_4^+ , NO_2^- , NO_3^- and total
206 suspended solids (TSS) (Table S2). Bromide (LOQ = 50 µg/L) and bromate (LOQ = 5 µg/L)
207 concentrations were measured for the first experiment after filtration on 0.7 µm GF/F glass filters
208 (Whatman) by ion chromatography (CARSO laboratory, Vénissieux, France).

209 Spectroscopic measurements were performed at room temperature directly after sampling and
210 filtration on 0.45 µm polyvinylidene fluoride filters (Milex[®], Merck). UV_{254} was measured in a 1
211 cm quartz cuvette using a spectrophotometer (Genesys 10S UV-Vis, Thermo Scientific).

212 Fluorescence emission/excitation matrices (EEMs) were obtained using a spectrofluorometer (FP-
213 8300, 150-W Xenon lamp, Jasco) with excitation wavelengths ranging from 240 nm to 450 nm at
214 an interval of 5 nm and emission wavelengths from 250 nm to 600 nm at an interval of 2 nm. The
215 samples were diluted with ultrapure water if the UV_{254} was higher than 0.080 1/cm to avoid inner
216 filter effects due to adsorbing species present in the wastewater (Alberts and Takács, 2004). EEMs
217 of ultrapure water were subtracted from the sample EEMs, which were also normalized by the
218 Raman peak of ultrapure water (excitation wavelength of 350 nm), to obtain fluorescence data in
219 Raman units (Lawaetz and Stedmon, 2009). A modified peak-picking approach based on a set of
220 six fluorescence indexes (excitation/emission wavelengths) was applied on the corrected EEMs.
221 These fluorescence indexes are based on maximum localization of 6 specific PARAFAC
222 components (C1-C6), similarly to the studies of Murphy et al. (2011) and Sgroi et al. (2017), that
223 were previously obtained from wastewater samples of the Seine Centre WWTP (Goffin et al.
224 2018). The C1 index (275/304 nm) is related to tyrosine-like compounds, C2 (285/344 nm) to
225 tyrosine/tryptophan-like compounds, C3 (275/378 nm) to tryptophan-like compounds, C4
226 (295/404 nm) to humic-like compounds from a recent biological activity and C5 (360/434 nm)
227 and C6 (275/464 nm) to humic-like compounds.

228 **3. Results and discussion**

229 *3.1. Influence of the specific ozone dose*

230 3.1.1. DOM removal

231 DOC removal globally increased with an increase of the specific ozone dose when TSS
232 concentration was below LOQ (experiment 1): < 5, 5, 14 and 17% at 0.18, 0.32, 0.67 and 0.97
233 $gO_3/gDOC$, respectively. When TSS were detected (2-7 mg/L), DOC removal was negligible (i.e.
234 < 5%) at 0.94 and 1.29 $gO_3/gDOC$ (experiment 5), and was even negative during experiment 2
235 with removals about -10, -25, and -14% for doses of 0.24, 0.60 and 0.90 $gO_3/gDOC$, respectively

236 (Table 2). The reaction of suspended solids (7 mg/L) with ozone during experiment 2 likely
237 converted particulate organic matter contained in TSS to DOC, as suggested by the decrease in
238 TSS concentration (from 7 to 3 mg/L) while TOC remained stable (~ 7 mgC/L). The effect of TSS
239 was also clearly observed during experiment 5 (TSS = 4 mg/L), exhibiting almost no removal of
240 DOC (< 5%) at 0.94 and 1.29 gO₃/gDOC, respectively, while TOC and TSS concentrations
241 decreased from 6 to ~ 4.5 mgC/L and 4 to 2 mg/L, respectively. Similar effects of suspended solids
242 on DOC removal by ozonation were observed in other studies (Jekel, 1994; Zhang et al., 2018).

243 Unlike DOC, the effective destruction of the unsaturated structures during ozonation, caused by
244 the electrophilic addition of ozone, lead to a regular UV₂₅₄ decrease with the specific ozone dose
245 (~ 10 to ~ 50%), but no effect of the TSS concentration was noted (Table 2). The absence of
246 plateau at high ozone doses was attributed to aromatic molecules (i.e. absorbing at 254 nm) present
247 in important quantity in the nitrified water. All WWTP-specific fluorescence indexes strongly
248 reacted with ozone, exhibiting a regular loss of intensity in all regions of EEMs (Figure S2) at
249 increasing ozone doses (Table 2). The loss of intensity was > 70% for specific ozone doses from
250 0.32 gO₃/gDOC and higher. At low specific ozone doses (0.18 and 0.24 gO₃/gDOC), tyrosine-like
251 organic compounds (C1 index) were less removed than other fluorescing DOM components, while
252 humic-like compounds (C5 index) were better removed at a dose of 0.18 gO₃/gDOC than other
253 components. The high reactivity of humic-like substances would be due to their high aromaticity,
254 conferring a high electron donating capacity and hence a high extent of the ozone-reactive sites.
255 Those results suggest a different reactivity of the various DOM components at low ozone doses
256 but a high removal of all fluorescing compounds at higher doses. Fluorescing moieties can
257 therefore be considered more reactive toward ozone than the DOM moieties absorbing at 254 nm.

258

259

260 Table 2. DOC, TSS and NO_2^- levels in the nitrified water before ozonation and removal of DOC, UV_{254}
 261 and 6 fluorescence WWTP-specific indexes (C1-C6) obtained from PARAFAC analysis (Goffin et al.
 262 2018) for different specific ozone doses.

Specific ozone dose (gO ₃ /gDOC)	Experiment	Nitrified water global parameters				Removal (%)							
		TSS (mg/L)	TOC (mgC/L)	DOC (mgC/L)	NO ₂ ⁻ (mg/L)	DOC	UV ₂₅₄	C1	C2	C3	C4	C5	C6
0.18	1	< 2	6	5.9	< 0.06	< 5	3	25	32	33	35	43	27
0.32						5	18	79	79	79	80	82	73
0.67						14	37	91	91	90	91	90	83
0.97						17	49	98	96	95	95	94	90
0.24	2	7	7.2	5.9	0.49	-10	15	25	42	45	41	43	35
0.60						-25	36	64	77	75	73	73	62
0.90						-14	49	91	91	89	87	85	77
0.24	3	< 2	5.1	4.8	< 0.06	< 5	15	34	39	37	36	44	28
0.09	4	< 2	4.6	4.5	0.26	< 5	12	-66	9	9	20	29	13
0.22						< 5	19	24	42	44	41	45	30
0.94	5	4	6.1	4.5	< 0.06	< 5	46	90	91	89	86	84	81
1.29						< 5	54	94	95	93	91	89	87

263 3.1.2. Bromate formation

264 The concentration of bromide ion in the nitrified water was 90 $\mu\text{g/L}$ and did not vary significantly
 265 after ozonation at all specific ozone doses used in experiment 1 (0.18-0.97 gO₃/gDOC). No
 266 bromate ion was detected at any specific ozone doses (i.e. < 5 $\mu\text{g/L}$) despite bromide levels in the
 267 range 50-100 $\mu\text{g/L}$ where bromate formation may become a problem (von Gunten, 2003b). The
 268 absence of bromate formation can be explained by the low reaction rate constant of bromide ion
 269 with ozone (160 L.mol⁻¹.s⁻¹, von Gunten, 2003b) in comparison to more reactive compounds such
 270 as DOM and OMPs. Zimmermann et al. (2011) observed bromate formation only at specific ozone
 271 doses higher than 0.90 gO₃/gDOC. The authors also observed NDMA formation at a specific
 272 ozone dose of 1.24 gO₃/gDOC and no N-nitrosomorpholine (NMOR). Contrary to bromate for
 273 which formation is mostly related to the specific ozone dose, NDMA formation rather depends on
 274 the concentrations and the conversion rate of precursors as well as matrix composition (Hollender
 275 et al., 2009). N-nitrosamines were not monitored in the study but future work will investigate their
 276 occurrence in similar conditions.

277

278 3.1.3. OMPs removal

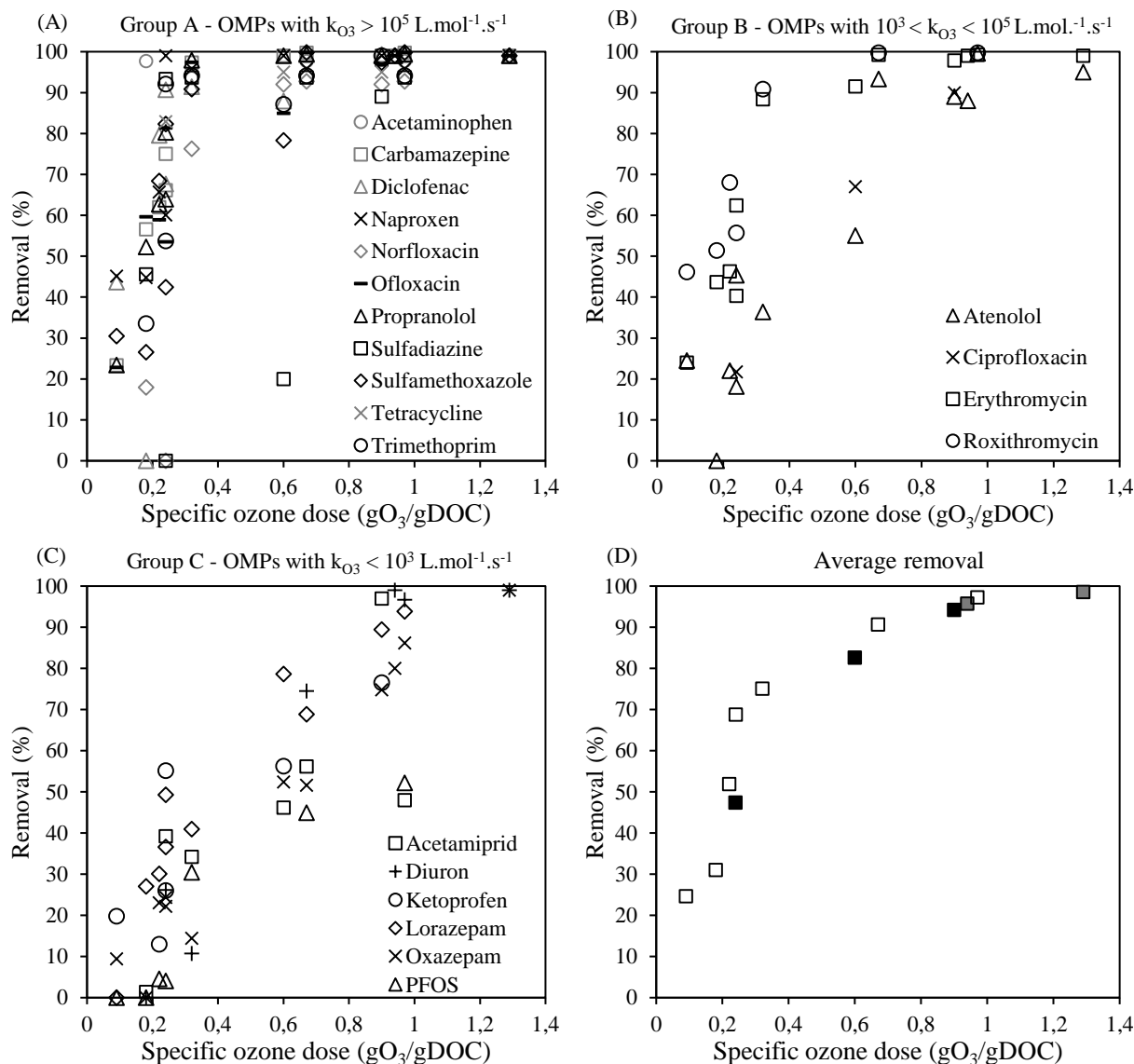
279 Among the 28 analysed compounds, 7 OMPs were never quantified in the nitrified water
280 (clothianidin, estradiol, estrone, ibuprofen, imidacloprid, thiachlopride and thiamethoxam), 11
281 OMPs were occasionally detected (acetaminophen, acetamiprid, ciprofloxacin, diuron,
282 ketoprofen, norfloxacin, PFOS, roxithromycin, sulfadiazine, tetracycline and trimethoprim) and
283 10 OMPs were always present (atenolol, carbamazepine, diclofenac, erythromycin, lorazepam,
284 naproxen, ofloxacin, oxazepam, propranolol and sulfamethoxazole) for each experiment (Table
285 S3). Removals of the quantified OMPs were calculated at each specific ozone dose employed
286 (Figure 1). Negative removals were obtained for diuron and PFOS at low specific ozone doses (<
287 0.30 gO₃/gDOC). They were attributed to matrix effects during analyses or to the formation of
288 these molecules after ozonation of metabolites/parent molecules. However, these negative
289 removals were low (-10 to 0%) and were not subsequently considered. As proposed by Margot et
290 al. (2013), OMPs were categorized in 3 groups depending on their expected reactivity and
291 observed removals (Table S4). Group A contained 11 OMPs (acetaminophen, carbamazepine,
292 diclofenac, naproxen, norfloxacin, ofloxacin, propranolol, sulfadiazine, sulfamethoxazole,
293 tetracycline and trimethoprim) exhibiting removals higher than 80% at specific ozone doses higher
294 than 0.30 gO₃/gDOC (Figure 1.A). These results were in agreement with reaction rate constants
295 from the literature, compounds from group A being rapidly oxidized by ozone ($k_{O_3} > 1 \times 10^5 \text{ L} \cdot \text{mol}^{-1} \cdot \text{s}^{-1}$).

297 4 OMPs (atenolol, ciprofloxacin, erythromycin and roxithromycin, listed as group B) were
298 supposed to react moderately with ozone ($1 \times 10^3 \text{ L} \cdot \text{mol}^{-1} \cdot \text{s}^{-1} < k_{O_3} < 1 \times 10^5 \text{ L} \cdot \text{mol}^{-1} \cdot \text{s}^{-1}$). Atenolol
299 and ciprofloxacin did react poorly with ozone at specific doses lower than 0.50 gO₃/gDOC
300 (removals < 40%) while high removals (> 80 %) were achieved for erythromycin and
301 roxithromycin for a dose of 0.32 gO₃/gDOC, similarly to group A (Figure 1.B). Erythromycin and
302 roxithromycin have a higher ozonation constant rate ($7 \times 10^4 \text{ L} \cdot \text{mol}^{-1} \cdot \text{s}^{-1}$) than atenolol and

303 ciprofloxacin ($2 \times 10^3 \text{ L}\cdot\text{mol}^{-1}\cdot\text{s}^{-1}$ and $2 \times 10^4 \text{ L}\cdot\text{mol}^{-1}\cdot\text{s}^{-1}$, respectively). A higher reactivity of
304 erythromycin and roxithromycin with hydroxyl radicals may also explain their higher removal.
305 DOM is rich in humic substances that can act either as radical scavenger or help in the formation
306 of hydroxyl radicals (Lee et al., 2013). The removal of atenolol and ciprofloxacin only reached
307 90% at specific ozone doses higher than $0.80 \text{ gO}_3/\text{gDOC}$.

308 Finally, group C comprised 4 OMPs (diuron, ketoprofen, oxazepam and PFOS) that were
309 demonstrated to be recalcitrant to ozone ($k_{\text{O}_3} < 1 \times 10^3 \text{ L}\cdot\text{mol}^{-1}\cdot\text{s}^{-1}$) and 2 OMPs (acetamiprid and
310 lorazepam) which ozone rate constants were not found in the literature but exhibited low removals
311 ($< 50\%$ at doses $< 0.5 \text{ gO}_3/\text{gDOC}$). High specific ozone doses ($> 0.90 \text{ gO}_3/\text{gDOC}$) allowed to
312 achieve high removals ($> 80\%$) for most OMPs of group C, but low removals were still obtained
313 for acetamiprid and PFOS (48% and 52% at $0.97 \text{ gO}_3/\text{gDOC}$, respectively).

314 The average removal at each specific ozone dose was determined based on the removals of the 10
315 OMPs systematically quantified in each experiment (Figure 1.D). The average removal was low
316 ($< 40\%$) at the lowest specific ozone doses and increased up to 90% with the higher specific dose.
317 The removal of many OMPs from groups A and B (i.e. 13 compounds) reached a plateau above
318 $0.30 \text{ gO}_3/\text{gDOC}$ ($> 80\%$). Going further this specific ozone dose only provided major gains of
319 removal for the remaining 8 OMPs (atenolol and ciprofloxacin and group C). A high variability
320 was observed at a specific ozone dose of $0.24 \text{ gO}_3/\text{gDOC}$ showing average removals of 47 and 69
321 % in the presence and absence of TSS, respectively. This difference can be explained by the TSS
322 levels during experiment 2. As proposed above, ozone probably reacted with the TSS which
323 decreased the quantity of ozone available to react with OMPs. The degradation of all OMPs was
324 affected at this specific ozone doses by TSS, especially naproxen (-39%), sulfadiazine (-93%),
325 sulfamethoxazole (-40%) and trimethoprim (-38%). Such a negative effect of suspended solids on
326 OMP removal from wastewater effluents by ozonation was also reported elsewhere (Zucker et al.,
327 2015), but the reason why some molecules are more impacted than others is not yet known.



328

329

330 Figure 1. Removals of the quantified OMPs (A, B and C) and average removal (D) at different specific
 331 ozone doses (white squares TSS = 2 mg/L, grey squares TSS = 4 mg/L, black squares TSS = 7 mg/L). The
 332 average removal was calculated from the removals of 10 OMPs that were systematically detected in each
 333 experiment. A table with detailed results for OMPs removal is presented in supplementary information
 334 (Table S4).

335 3.2. Influence of the activated carbon dose

336 3.2.1. DOM removal

337 Before adsorption, the nitrified water was ozonated at a dose of 0.24 gO₃/gDOC (experiment 3).

338 The removal of DOC, UV₂₅₄ and fluorescence WWTP-specific indexes increased with the PAC

339 dose (Table S5). At the lowest PAC dose of 2 mg/L, all removals were low and similar (0-6 %),

340 the highest removal being attained for the C2 index related to tyrosine and tryptophan-like

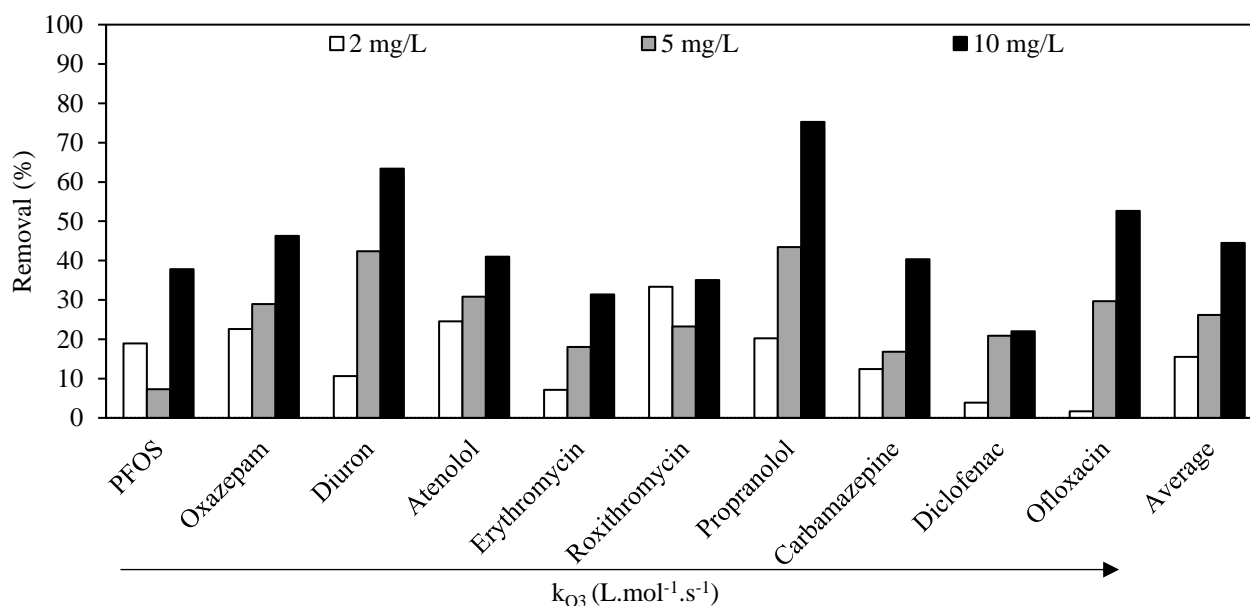
341 compounds. At a PAC dose of 5 mg/L, the C2 index exhibited again the highest removal (20%)

342 and other removals ranged between 1 and 11%. A low removal was observed for UV₂₅₄ with a
343 PAC dose of 10 mg/L, while the best removals were obtained for indexes C1, C2, C3 and C4 (18-
344 23%). These 4 indexes are related to small DOM fractions (aromatic proteins-like, tyrosine-like,
345 tryptophan-like and humic-like from a recent biological activity) which had probably a better
346 access to the PAC porosity in 30 min. Sgroi et al. (2018) observed a better removal of humic-like
347 fractions than tyrosine and tryptophan-like, but it is well known that that the distribution of
348 activated carbon porosity between macro-, meso- and micropores has an influence on DOM
349 adsorption (Guillossou et al., 2020; Li et al., 2003). As a result, the different type of activated
350 carbon used in this study led to a different behavior toward the adsorption of various DOM
351 fractions.

352 3.2.2. OMPs removal

353 Among the 28 compounds analyzed, 10 OMPs (acetaminophen, acetamiprid, clothianidin,
354 estradiol, ibuprofen, imidacloprid, norfloxacin, tetracycline, thiacloprid and thiamethoxam) were
355 not quantified in the ozonated water (0.24 gO₃/gDOC, experiment 3). 8 compounds (ciprofloxacin,
356 estrone, ketoprofen, lorazepam, naproxen, sulfadiazine, sulfamethoxazole and trimethoprim) were
357 quantified at concentrations close to their LOQ so their removal was not calculated. Removals of
358 the 10 remaining OMPs at the 3 different PAC doses (2, 5 and 10 mg/L) are presented in Figure
359 2.

360



361
 362 Figure 2. Removals of the 10 OMPs quantified in the ozonated effluent (0.24 gO₃/gDOC) and the average
 363 removal at 3 different PAC doses (2, 5 and 10 mg/L) after 30 min of contact time (experiment 3). OMPs
 364 are ranked with respect to their k_{O₃} value (increasing from left to right).

365 OMPs removals were all lower than 30% at a PAC dose of 2 mg/L. At 5 mg/L of PAC, removals
 366 remained between 10 and 30% except for diuron (42%) and propranolol (43%). Those 2
 367 compounds showed a better adsorption than other OMPs in other studies (Margot et al., 2013;
 368 Ruel et al., 2011). In a previous study using PAC, their removal was higher than 60% at a dose of
 369 10 mg/L while those of other OMPs remained between 20 and 60% (Mailler et al., 2015). More
 370 information on the effect of the physico-chemical properties of OMPs on their adsorption can be
 371 found in a previous study (Guillossou et al., 2020). The average removal increased regularly with
 372 the dose (14, 25 and 42%). A PAC dose of 10 mg/L provided the highest removals after ozonation
 373 and was thus selected for the subsequent experiments. Notably, substantial removals (> 30%) were
 374 obtained for compounds presenting a low k_{O₃} (e.g. PFOS, oxazepam, diuron) which would
 375 guarantee their removal during a coupled ozonation/adsorption process.

376 3.3. Influence of ozonation on adsorption performances

377 3.3.1. Influence of ozonation on DOM adsorption

378 DOC and UV removals by adsorption after ozonation at low specific doses (0.09 and 0.22
 379 gO₃/gDOC, experiment 4) remained low (< 12%) and were not significantly different from

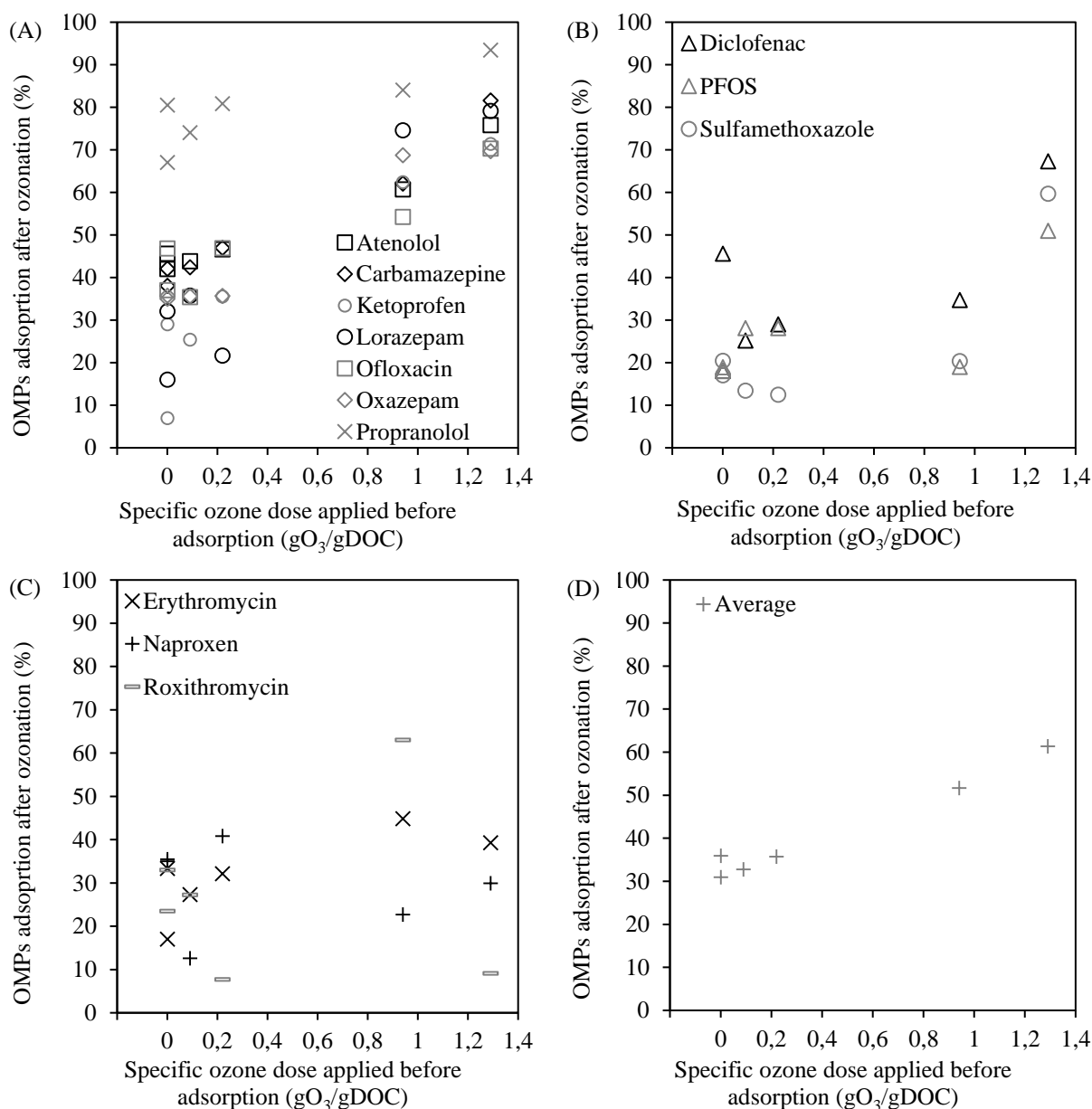
380 nitrified water, while a slight increase was observed for fluorescence indexes (Table S6).
381 However, the removal of DOC, UV_{254} and all fluorescence indexes by adsorption after ozonation
382 at 0.94 and 1.29 $gO_3/gDOC$ (experiment 5) decreased, which suggests that DOM adsorbability
383 decreased at high specific ozone doses. These results are in accordance with the observations of
384 Zietzschmann et al. (2015) who found a decrease in DOM adsorbability at high specific ozone
385 consumptions (0.65 and 1.64 $gO_3/gDOC$) due to a reduction of DOM aromaticity, molecular size
386 and hydrophobicity.

387 3.3.2. Influence of ozonation on OMPs adsorption

388 Removals by adsorption onto PAC (10 mg/L) after ozonation at 4 different specific ozone doses
389 were obtained for the 13 OMPs quantified before adsorption (Figure 3). The average removal after
390 adsorption was similar between the nitrified water (31-36%) and the ozonated water at low
391 specific ozone doses (33 and 36% at 0.09 and 0.22 $gO_3/gDOC$, respectively, experiment 4)
392 suggesting no effect of pre-ozonation on OMP adsorption at these specific doses.

393 OMPs adsorption increased at higher specific ozone doses (experiment 5) with an average removal
394 of 52% for a dose of 0.94 $gO_3/gDOC$ and 61% for a dose of 1.29 $gO_3/gDOC$, confirming the
395 positive effect of high ozone doses ($> 0.50 gO_3/gDOC$) on the subsequent adsorption, as
396 previously suggested by Zietzschmann et al. (2015). The decrease in DOM adsorbability (also
397 observed in the study, see part 3.3.1) probably involves a reduction of the competitive adsorption
398 with OMPs. Individual removals of atenolol, carbamazepine, ketoprofen, lorazepam, ofloxacin,
399 oxazepam and propranolol (Figure 3.A) followed a similar behavior to the average removal
400 (Figure 3.D). For some OMPs (diclofenac, PFOS and sulfamethoxazole), the increase of their
401 removal was only observed at the highest specific ozone dose (1.29 $gO_3/gDOC$), while ozonation
402 at any dose had no effect on the adsorption of erythromycin, naproxen and roxithromycin. The
403 adsorption of erythromycin and roxithromycin was previously demonstrated to be impacted by

404 steric hindrance because of their high molecular weight, and thus are not subject to competition
 405 with DOM (Guillossou et al., 2020).



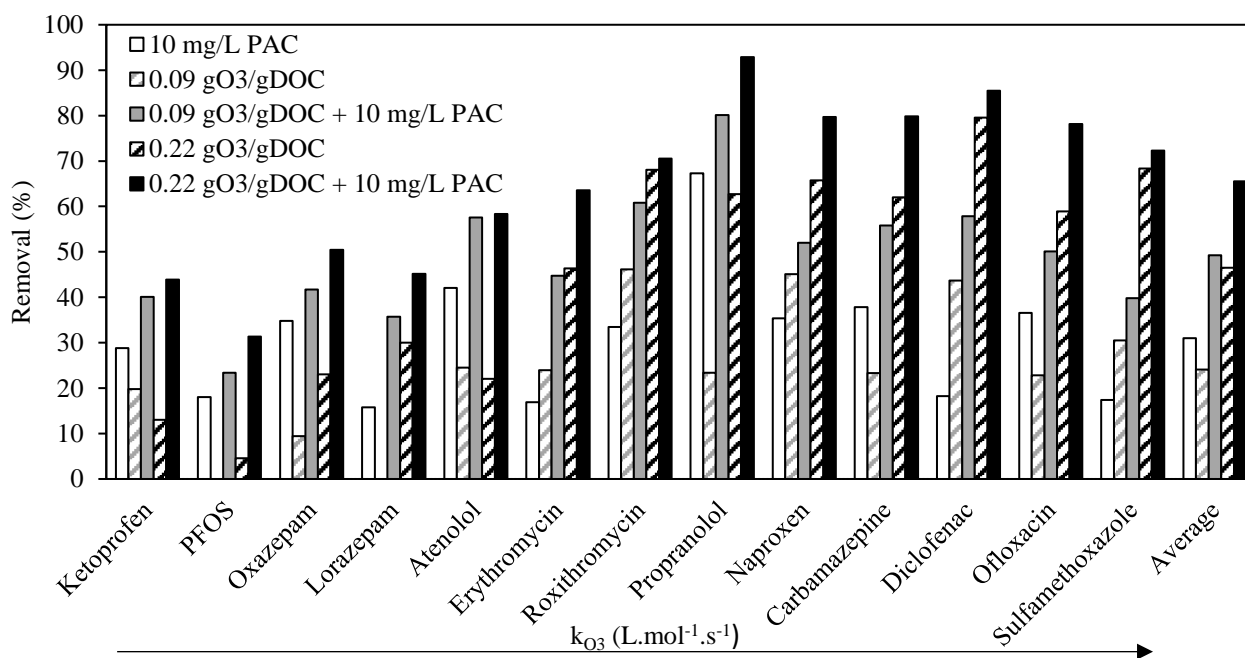
407
 408 Figure 3. OMPs removal by adsorption onto PAC (10 mg/L) after ozonation of the nitrified water at various
 409 specific ozone doses.

410 3.4. Comparison of coupled ozonation/adsorption with adsorption alone

411 3.4.1. OMPs removal

412 Among the 28 analyzed compounds, 7 OMPs (acetamiprid, clothianidin, estradiol, estrone,
 413 imidacloprid, thiacloprid and thiamethoxam) were not quantified in the nitrified water during

414 experiment 4 (Table S3). In addition, the removal of 8 OMPs (acetaminophen, ciprofloxacin,
 415 diuron, estrone, ibuprofen, norfloxacin, sulfadiazine and trimethoprim) was not calculated
 416 because their concentrations in the nitrified water were close to their respective LOQ. As expected,
 417 higher overall removals were obtained with ozonation coupled to adsorption (average removal of
 418 49 and 66% with 0.09 and 0.22 gO₃/gDOC + 10 mg/L of PAC, respectively) than the sole
 419 adsorption (average removal of 31% with a PAC dose of 10 mg/L) (Figure 4). The highest
 420 removals were achieved when combining 0.22 gO₃/gDOC and 10 mg/L of PAC, reaching more
 421 than 80% for 4 OMPs (carbamazepine, diclofenac, naproxen and propranolol). 5 OMPs exhibited
 422 moderate removals ranging from 58 to 72% (atenolol, erythromycin, ofloxacin, roxithromycin
 423 and sulfamethoxazole) and 4 OMPs were less removed than the average with removals ranging
 424 from 31 to 50% (ketoprofen, lorazepam, oxazepam and PFOS). At higher ozone doses (0.94 and
 425 1.29 gO₃/gDOC, experiment 5), all OMPs reached removals > 99%, except atenolol and oxazepam
 426 with 88 and 80% removal at 0.94 gO₃/gDOC (data not shown on Figure 4).



427
 428 Figure 4. Removals of the 13 quantified OMPs and the average removal at different ozone and PAC doses (experiment
 429 4). OMPs are ranked with respect to their k_{O3} value (increasing from left to right).

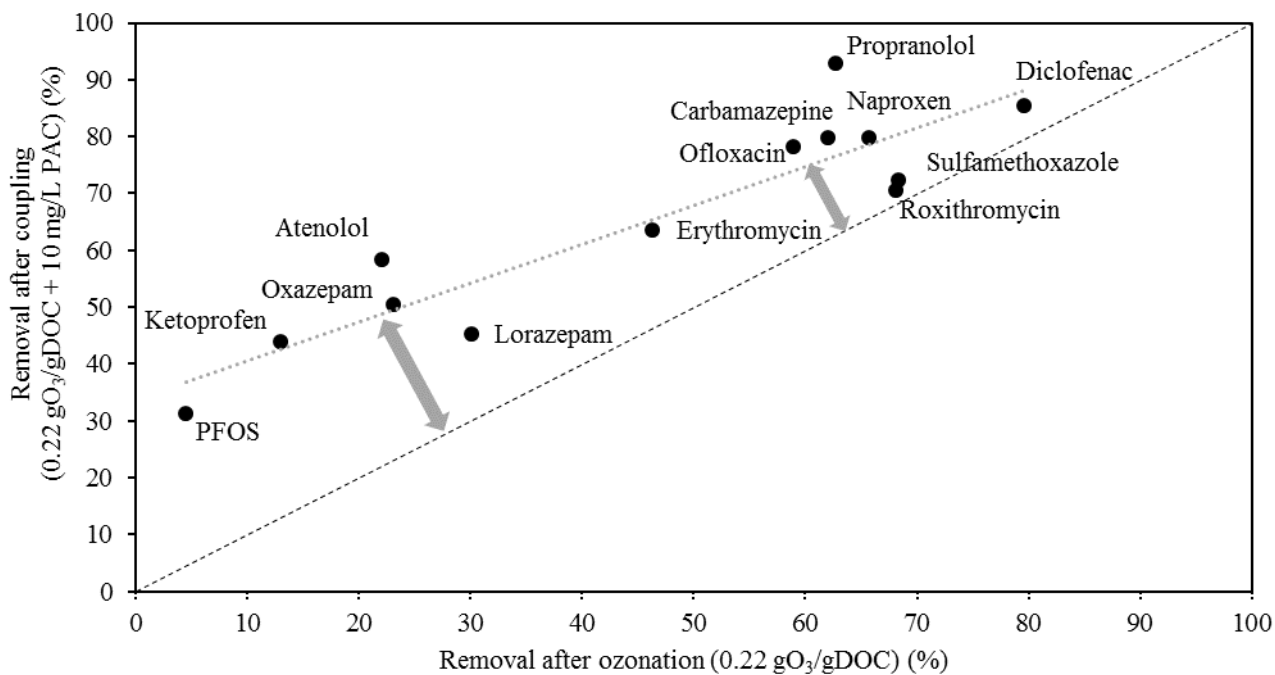
430
 431
 432

433 3.4.2. Complementarity of the two processes

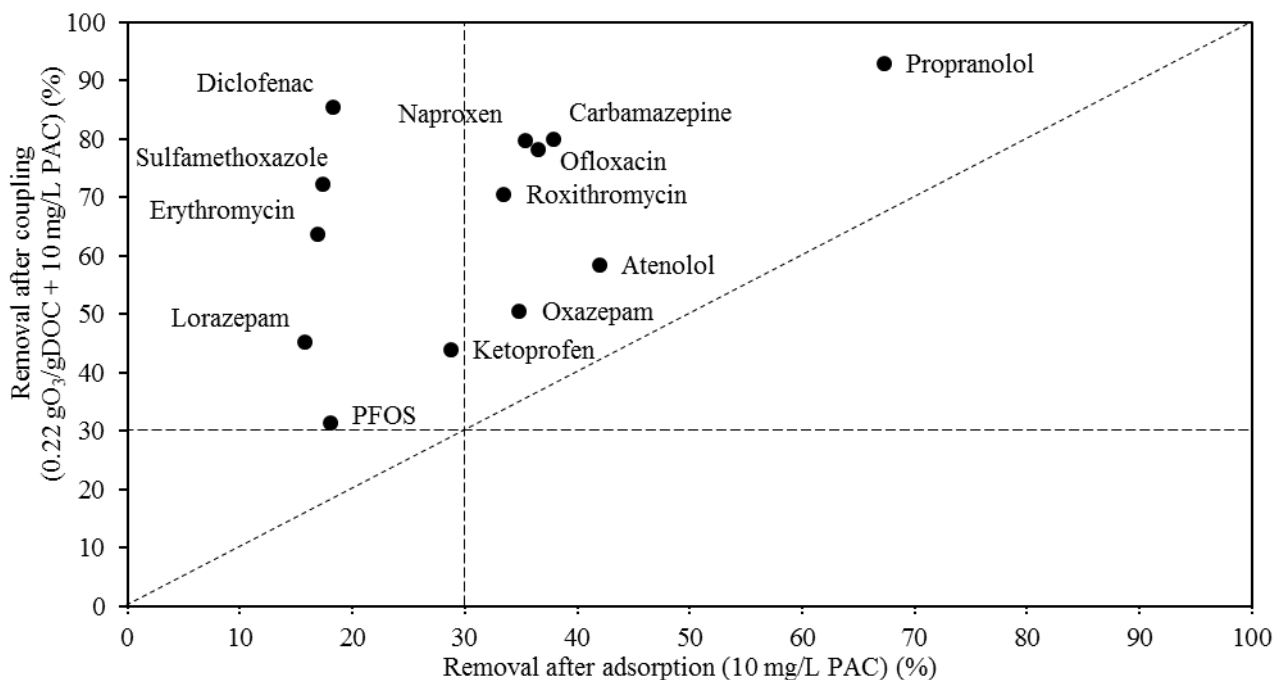
434 As illustrated in Figure 5, a general increase in removals was provided by the coupling of
435 ozonation with adsorption, especially for molecules that were not well removed by ozonation
436 alone. After ozonation at 0.22 gO₃/gDOC (experiment 4), PAC adsorption improved the removal
437 of OMPs with low k_{O₃} values (e.g. ketoprofen, oxazepam, PFOS, k_{O₃} < 10⁴ L.mol⁻¹.s⁻¹) with an
438 additional removal of 25% in average. OMPs with high k_{O₃} values (e.g. diclofenac, naproxen,
439 ofloxacin, k_{O₃} > 10⁴ L.mol⁻¹.s⁻¹) exhibited an additional removal by adsorption of only 12% in
440 average. At the lower ozone dose of 0.09 gO₃/gDOC (experiment 4), overall removals of
441 compounds reacting slowly with ozone were also higher (additional removal by PAC of 28% in
442 average) as compared to the ozone-reactive OMPs (additional removal by PAC of 22% in average)
443 (Figure S3). At higher ozone doses (0.94 and 1.29 gO₃/gDOC, experiment 5), the removal of
444 OMPs by ozone was largely predominant (> 99% for almost all molecules), so the gain provided
445 by the adsorption was insignificant (data not shown).

446 Conversely, ozonation coupled to activated carbon adsorption was also interesting for OMPs
447 exhibiting a low affinity with PAC such as diclofenac, erythromycin, lorazepam, PFOS and
448 sulfamethoxazole (Figure 6 and Figure S4). The addition of an ozonation step before adsorption
449 provided an additional removal, for OMPs refractory to adsorption (i.e. removal < 30% for the
450 sole adsorption), of 21 and 38% in average for specific ozone doses of 0.09 and 0.22 gO₃/gDOC,
451 respectively. For compounds that were better adsorbed such as atenolol, carbamazepine,
452 ketoprofen, naproxen, ofloxacin, oxazepam, propranolol and roxithromycin (removals > 30% by
453 PAC), the additional removal provided by ozonation was lower but still significant (16 and 32%
454 in average at 0.09 and 0.22 gO₃/gDOC, respectively). The removal of all the quantified OMPs
455 was higher than 30% after ozonation and adsorption. These results are particularly interesting
456 because they highlight a possible optimum in the operating conditions when ozonation is
457 combined with activated carbon adsorption to treat OMPs in real wastewater effluents. In addition,

458 the use of activated carbon should allow the adsorption of ozonation byproducts formed during
 459 ozonation (e.g. hydroxylated products or N-oxides) (Merel et al., 2017). Additional studies should
 460 be conducted (e.g. by using non-targeted analyses, Schollée et al. (2018)) to study if such
 461 transformation products can be effectively adsorbed on activated carbon.



462
 463 Figure 5. OMPs removal after coupling (0.22 gO₃/gDOC + 10 mg/L PAC) compared to ozonation alone
 464 (0.22 gO₃/gDOC).



465
 466 Figure 6. OMPs removal after coupling (0.22 gO₃/gDOC + 10 mg/L PAC) compared to adsorption alone
 467 (10 mg/L).

468

469 **Conclusion**

- 470 • Ozonation is an efficient process for OMPs elimination from wastewater effluents, with
471 average removals higher than 80% for specific ozone doses higher than 0.60 gO₃/gDOC.
472 An increase in the average removal was observed with the increase of the specific ozone
473 dose and the removals of 13 reactive compounds reached a plateau above 0.30 gO₃/gDOC.
474 Higher doses (> 0.80 gO₃/gDOC) were necessary for less reactive compounds.
- 475 • The ozonation performances for OMPs removal thus seemed to be dependent of the
476 wastewater quality, especially for low specific ozone doses where a higher competition
477 between OMPs and the matrix for ozone can be expected. Online monitoring of the quality
478 of the matrix (DOC, TSS, NO₂⁻) therefore seems interesting to achieve stable ozonation
479 performances. Further study is needed to better understand the negative influence of TSS
480 on ozonation performances and the conversion of TSS to DOC. A correction of the specific
481 ozone dose according to the TSS concentration, as for nitrites, could then be considered.
482 No bromate was detected at any specific ozone doses in this study.
- 483 • The influence of ozonation on adsorption performances was not significant at low specific
484 ozone doses (0.09 and 0.22 gO₃/gDOC) but an increase in OMPs adsorption was observed
485 at higher doses (0.94 and 1.29 gO₃/gDOC). This increase was attributed to the decrease in
486 DOM adsorbability which led to a reduced adsorption competition with OMPs. Further
487 experiments are needed to determine precisely the specific ozone dose at which the
488 adsorbability of DOM decreases and to identify which DOM fractions are responsible for
489 this decrease after oxidation.
- 490 • A better adsorption of OMPs that were poorly reactive toward ozone was observed as
491 compared to the adsorption of highly ozone-reactive OMPs. Conversely, OMPs refractory
492 to adsorption were well removed by ozonation. As a result, the removal of all OMPs were
493 substantially improved with the addition of ozonation prior to adsorption. Even if low

494 specific ozone doses did not improve OMPs adsorption in comparison to higher doses, low
495 ozone doses coupled to activated carbon adsorption appear as complementary and an
496 adequate process for the removal of various micropollutants in wastewater effluents. The
497 formation and adsorption of ozonation transformation products should also be carried out
498 to better evaluate the complementarity of the two processes. A life cycle analysis is needed
499 to estimate the benefit of reducing ozone doses in terms of energy consumption while
500 using a subsequent adsorption onto activated carbon as compared to a single ozonation
501 step at higher doses coupled to adsorption or sand filtration.

- 502 • Based on the quality of the wastewater studied, a specific ozone dose of 0.20-0.30
503 gO_3/gDOC and a PAC dose of 10 mg/L is recommended to achieve satisfactory
504 performances for a wide range of OMPs (e.g. a removal higher than 80% for
505 carbamazepine and diclofenac which are used as indicator substances in the Swiss
506 regulation).

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512 **References**

- 513 Alberts, J.J., Takács, M., 2004. Comparison of the natural fluorescence distribution among size
514 fractions of terrestrial fulvic and humic acids and aquatic natural organic matter. *Org.*
515 *Geochem.* 35, 1141–1149. <https://doi.org/10.1016/j.orggeochem.2004.06.010>
- 516 Altmann, J., Ruhl, A.S., Sauter, D., Pohl, J., Jekel, M., 2015a. How to dose powdered activated
517 carbon in deep bed filtration for efficient micropollutant removal. *Water Res.* 78, 9–17.
518 <https://doi.org/10.1016/j.watres.2015.03.031>
- 519 Altmann, J., Sperlich, A., Jekel, M., 2015b. Integrating organic micropollutant removal into
520 tertiary filtration: Combining PAC adsorption with advanced phosphorus removal. *Water*
521 *Res.* 84, 58–65. <https://doi.org/10.1016/j.watres.2015.07.023>

- 522 Alves, T.C., Cabrera-Codony, A., Barceló, D., Rodriguez-Mozaz, S., Pinheiro, A., Gonzalez-
523 Olmos, R., 2018. Influencing factors on the removal of pharmaceuticals from water with
524 micro-grain activated carbon. *Water Res.* <https://doi.org/10.1016/j.watres.2018.07.037>
- 525 Arias Espana, V.A., Mallavarapu, M., Naidu, R., 2015. Treatment technologies for aqueous
526 perfluorooctanesulfonate (PFOS) and perfluorooctanoate (PFOA): A critical review with
527 an emphasis on field testing. *Environ. Technol. Innov.* 4, 168–181.
528 <https://doi.org/10.1016/j.eti.2015.06.001>
- 529 Benstoem, F., Nahrstedt, A., Boehler, M., Knopp, G., Montag, D., Siegrist, H., Pinnekamp, J.,
530 2017. Performance of granular activated carbon to remove micropollutants from municipal
531 wastewater—A meta-analysis of pilot- and large-scale studies. *Chemosphere* 185, 105–
532 118. <https://doi.org/10.1016/j.chemosphere.2017.06.118>
- 533 Bourgin, M., Beck, B., Boehler, M., Borowska, E., Fleiner, J., Salhi, E., Teichler, R., von Gunten,
534 U., Siegrist, H., McArdell, C.S., 2018. Evaluation of a full-scale wastewater treatment
535 plant upgraded with ozonation and biological post-treatments: Abatement of
536 micropollutants, formation of transformation products and oxidation by-products. *Water*
537 *Res.* 129, 486–498. <https://doi.org/10.1016/j.watres.2017.10.036>
- 538 Choubert, J.-M., Martin Ruel, S., Miege, C., Coquery, M., 2017. Rethinking micropollutant
539 removal assessment methods for wastewater treatment plants – how to get more robust
540 data? *Water Sci. Technol.* 75, 2964–2972. <https://doi.org/10.2166/wst.2017.181>
- 541 Cruz-Alcalde, A., Sans, C., Esplugas, S., 2017. Priority pesticides abatement by advanced water
542 technologies: The case of acetamiprid removal by ozonation. *Sci. Total Environ.* 599–600,
543 1454–1461. <https://doi.org/10.1016/j.scitotenv.2017.05.065>
- 544 de Wilt, A., van Gijn, K., Verhoek, T., Vergnes, A., Hoek, M., Rijnaarts, H., Langenhoff, A.,
545 2018. Enhanced pharmaceutical removal from water in a three step bio-ozone-bio process.
546 *Water Res.* 138, 97–105. <https://doi.org/10.1016/j.watres.2018.03.028>
- 547 Deborde, M., Rabouan, S., Duguet, J.-P., Legube, B., 2005. Kinetics of Aqueous Ozone-Induced
548 Oxidation of Some Endocrine Disruptors. *Environ. Sci. Technol.* 39, 6086–6092.
549 <https://doi.org/10.1021/es0501619>
- 550 Dodd, M.C., Buffle, M.-O., von Gunten, U., 2006. Oxidation of Antibacterial Molecules by
551 Aqueous Ozone: Moiety-Specific Reaction Kinetics and Application to Ozone-Based
552 Wastewater Treatment. *Environ. Sci. Technol.* 40, 1969–1977.
553 <https://doi.org/10.1021/es051369x>
- 554 Garoma, T., Umamaheshwar, S.K., Mumper, A., 2010. Removal of sulfadiazine, sulfamethizole,
555 sulfamethoxazole, and sulfathiazole from aqueous solution by ozonation. *Chemosphere*
556 79, 814–820. <https://doi.org/10.1016/j.chemosphere.2010.02.060>
- 557 Goffin, A., Guérin, S., Rocher, V., Varrault, G., 2018. Towards a better control of the wastewater
558 treatment process: excitation-emission matrix fluorescence spectroscopy of dissolved
559 organic matter as a predictive tool of soluble BOD5 in influents of six Parisian wastewater
560 treatment plants. *Environ. Sci. Pollut. Res.* <https://doi.org/10.1007/s11356-018-1205-1>
- 561 Grover, D.P., Zhou, J.L., Frickers, P.E., Readman, J.W., 2011. Improved removal of estrogenic
562 and pharmaceutical compounds in sewage effluent by full scale granular activated carbon:
563 Impact on receiving river water. *J. Hazard. Mater.* 185, 1005–1011.
564 <https://doi.org/10.1016/j.jhazmat.2010.10.005>
- 565 Guilloso, R., Le Roux, J., Mailler, R., Morlay, C., Vulliet, E., Nauleau, F., Rocher, V., Gasperi,
566 J., 2020. Influence of the properties of 7 micro-grain activated carbons on organic
567 micropollutants removal from wastewater effluent. *Chemosphere* 243, 125306.
568 <https://doi.org/10.1016/j.chemosphere.2019.125306>
- 569 Guilloso, R., Le Roux, J., Mailler, R., Vulliet, E., Morlay, C., Nauleau, F., Gasperi, J., Rocher,
570 V., 2019. Organic micropollutants in a large wastewater treatment plant: What are the
571 benefits of an advanced treatment by activated carbon adsorption in comparison to

572 conventional treatment? *Chemosphere* 218, 1050–1060.
573 <https://doi.org/10.1016/j.chemosphere.2018.11.182>

574 Hamdi El Najjar, N., Touffet, A., Deborde, M., Journel, R., Karpel Vel Leitner, N., 2014. Kinetics
575 of paracetamol oxidation by ozone and hydroxyl radicals, formation of transformation
576 products and toxicity. *Sep. Purif. Technol.* 136, 137–143.
577 <https://doi.org/10.1016/j.seppur.2014.09.004>

578 Hollender, J., Zimmermann, S.G., Koepke, S., Krauss, M., McArdell, C.S., Ort, C., Singer, H.,
579 von Gunten, U., Siegrist, H., 2009. Elimination of Organic Micropollutants in a Municipal
580 Wastewater Treatment Plant Upgraded with a Full-Scale Post-Ozonation Followed by
581 Sand Filtration. *Environ. Sci. Technol.* 43, 7862–7869. <https://doi.org/10.1021/es9014629>

582 Huber, M.M., Canonica, S., Park, G.-Y., von Gunten, U., 2003. Oxidation of Pharmaceuticals
583 during Ozonation and Advanced Oxidation Processes. *Environ. Sci. Technol.* 37, 1016–
584 1024. <https://doi.org/10.1021/es025896h>

585 Huber, M.M., Göbel, A., Joss, A., Hermann, N., Löffler, D., McArdell, C.S., Ried, A., Siegrist,
586 H., Ternes, T.A., von Gunten, U., 2005. Oxidation of Pharmaceuticals during Ozonation
587 of Municipal Wastewater Effluents: A Pilot Study. *Environ. Sci. Technol.* 39, 4290–4299.
588 <https://doi.org/10.1021/es048396s>

589 Hübner, U., von Gunten, U., Jekel, M., 2015. Evaluation of the persistence of transformation
590 products from ozonation of trace organic compounds – A critical review. *Water Res.* 68,
591 150–170. <https://doi.org/10.1016/j.watres.2014.09.051>

592 Jekel, M.R., 1994. Flocculation Effects of Ozone. *Ozone Sci. Eng.* 16, 55–66.
593 <https://doi.org/10.1080/01919519408552380>

594 Kårelid, V., Larsson, G., Björleinius, B., 2017. Effects of recirculation in a three-tank pilot-scale
595 system for pharmaceutical removal with powdered activated carbon. *J. Environ. Manage.*
596 193, 163–171. <https://doi.org/10.1016/j.jenvman.2017.01.078>

597 Lawaetz, A.J., Stedmon, C.A., 2009. Fluorescence Intensity Calibration Using the Raman Scatter
598 Peak of Water. *Appl. Spectrosc.* 63, 936–940.

599 Lee, Y., Kovalova, L., McArdell, C.S., von Gunten, U., 2014. Prediction of micropollutant
600 elimination during ozonation of a hospital wastewater effluent. *Water Res.* 64, 134–148.
601 <https://doi.org/10.1016/j.watres.2014.06.027>

602 Li, Q., Snoeyink, V.L., Mariñas, B.J., Campos, C., 2003. Pore blockage effect of NOM on atrazine
603 adsorption kinetics of PAC: the roles of PAC pore size distribution and NOM molecular
604 weight. *Water Res.* 37, 4863–4872. <https://doi.org/10.1016/j.watres.2003.08.018>

605 Ling, W., Ben, W., Xu, K., Zhang, Y., Yang, M., Qiang, Z., 2018. Ozonation of norfloxacin and
606 levofloxacin in water: Specific reaction rate constants and defluorination reaction.
607 *Chemosphere* 195, 252–259. <https://doi.org/10.1016/j.chemosphere.2017.12.079>

608 Mailler, R., Gasperi, J., Coquet, Y., Buleté, A., Vulliet, E., Deshayes, S., Zedek, S., Mirande-Bret,
609 C., Eudes, V., Bressy, A., Caupos, E., Moilleron, R., Chebbo, G., Rocher, V., 2016a.
610 Removal of a wide range of emerging pollutants from wastewater treatment plant
611 discharges by micro-grain activated carbon in fluidized bed as tertiary treatment at large
612 pilot scale. *Sci. Total Environ.* 542, Part A, 983–996.
613 <http://dx.doi.org/10.1016/j.scitotenv.2015.10.153>

614 Mailler, R., Gasperi, J., Coquet, Y., Derome, C., Buleté, A., Vulliet, E., Bressy, A., Varrault, G.,
615 Chebbo, G., Rocher, V., 2016b. Removal of emerging micropollutants from wastewater
616 by activated carbon adsorption: Experimental study of different activated carbons and
617 factors influencing the adsorption of micropollutants in wastewater. *J. Environ. Chem.*
618 *Eng.* 4, 1102–1109. <https://doi.org/10.1016/j.jece.2016.01.018>

619 Mailler, R., Gasperi, J., Coquet, Y., Deshayes, S., Zedek, S., Cren-Olivé, C., Cartiser, N., Eudes,
620 V., Bressy, A., Caupos, E., Moilleron, R., Chebbo, G., Rocher, V., 2015. Study of a large
621 scale powdered activated carbon pilot: Removals of a wide range of emerging and priority

622 micropollutants from wastewater treatment plant effluents. *Water Res.* 72, 315–330.
623 <https://doi.org/10.1016/j.watres.2014.10.047>

624 Mailler, R., Gasperi, J., Rocher, V., Gilbert-Pawlik, S., Geara-Matta, D., Moilleron, R., Chebbo,
625 G., 2014. Biofiltration vs conventional activated sludge plants: what about priority and
626 emerging pollutants removal? *Environ. Sci. Pollut. Res.* 21, 5379–5390.
627 <https://doi.org/10.1007/s11356-013-2388-0>

628 Margot, J., Kienle, C., Magnet, A., Weil, M., Rossi, L., de Alencastro, L.F., Abegglen, C.,
629 Thonney, D., Chèvre, N., Schärer, M., Barry, D.A., 2013. Treatment of micropollutants in
630 municipal wastewater: Ozone or powdered activated carbon? *Sci. Total Environ.* 461–462,
631 480–498. <https://doi.org/10.1016/j.scitotenv.2013.05.034>

632 Márquez, G., Rodríguez, E.M., Beltrán, F.J., Álvarez, P.M., 2013. Determination of Rate
633 Constants for Ozonation of Ofloxacin in Aqueous Solution. *Ozone Sci. Eng.* 35, 186–195.
634 <https://doi.org/10.1080/01919512.2013.771530>

635 Mathon, B., Coquery, M., Miege, C., Penru, Y., Choubert, J.-M., 2017. Removal efficiencies and
636 kinetic rate constants of xenobiotics by ozonation in tertiary treatment. *Water Sci. Technol.*
637 75, 2737–2746. <https://doi.org/10.2166/wst.2017.114>

638 Matsui, Y., Yuasa, A., Li, F.-S., 1999. Pretreatment effects on activated carbon adsorption of
639 humic substances: Distributed fictive component analysis. *Water Sci. Technol.* 40, 223–
640 230. [https://doi.org/10.1016/S0273-1223\(99\)00660-5](https://doi.org/10.1016/S0273-1223(99)00660-5)

641 Meinel, F., Ruhl, A.S., Sperlich, A., Zietzschmann, F., Jekel, M., 2015. Pilot-Scale Investigation
642 of Micropollutant Removal with Granular and Powdered Activated Carbon. *Water. Air.*
643 *Soil Pollut.* 226. <https://doi.org/10.1007/s11270-014-2260-y>

644 Meinel, F., Zietzschmann, F., Ruhl, A.S., Sperlich, A., Jekel, M., 2016. The benefits of powdered
645 activated carbon recirculation for micropollutant removal in advanced wastewater
646 treatment. *Water Res.* 91, 97–103. <https://doi.org/10.1016/j.watres.2016.01.009>

647 Merel, S., Lege, S., Yanez Heras, J.E., Zwiener, C., 2017. Assessment of N-Oxide Formation
648 during Wastewater Ozonation. *Environ. Sci. Technol.* 51, 410–417.
649 <https://doi.org/10.1021/acs.est.6b02373>

650 Muller, U., Hess, F., Worch, E., 1996. Impact of organic matter adsorbability on micropollutant
651 removal by activated carbon. *J. WATER SUPPLY Res. Technol.-AQUA* 45, 273–280.

652 Murphy, K.R., Hambly, A., Singh, S., Henderson, R.K., Baker, A., Stuetz, R., Khan, S.J., 2011.
653 Organic Matter Fluorescence in Municipal Water Recycling Schemes: Toward a Unified
654 PARAFAC Model. *Environ. Sci. Technol.* 45, 2909–2916.
655 <https://doi.org/10.1021/es103015e>

656 Naumov, S., Mark, G., Jarocki, A., von Sonntag, C., 2010. The Reactions of Nitrite Ion with
657 Ozone in Aqueous Solution - New Experimental Data and Quantum-Chemical
658 Considerations. *OZONE-Sci. Eng.* 32, 430–434.
659 <https://doi.org/10.1080/01919512.2010.522960>

660 Newcombe, G., Morrison, J., Hepplewhite, C., Knappe, D.R.U., 2002. Simultaneous adsorption
661 of MIB and NOM onto activated carbon II. Competitive effects. *Carbon* 40, 2147–2156.

662 Office fédéral de l'environnement, Division Eaux, 2014. Rapport explicatif concernant la
663 modification de l'ordonnance sur la protection des eaux.

664 Real, F.J., Benitez, F.J., Acero, J.L., Sagasti, J.J.P., Casas, F., 2009. Kinetics of the Chemical
665 Oxidation of the Pharmaceuticals Primidone, Ketoprofen, and Diatrizoate in Ultrapure and
666 Natural Waters. *Ind. Eng. Chem. Res.* 48, 3380–3388. <https://doi.org/10.1021/ie801762p>

667 Rocher, V., Paffoni, C., Gonçalves, A., Guérin, S., Azimi, S., Gasperi, J., Moilleron, R., Pauss,
668 A., 2012. Municipal wastewater treatment by biofiltration: comparisons of various
669 treatment layouts. Part 1: assessment of carbon and nitrogen removal. *Water Sci. Technol.*
670 65, 1705. <https://doi.org/10.2166/wst.2012.105>

671 Roustan, M., Debellefontaine, H., Do-Quang, Z., Duguet, J.-P., 1998. Development of a Method
672 for the Determination of Ozone Demand of a Water. *Ozone Sci. Eng.* 20, 513–520.
673 <https://doi.org/10.1080/01919519809480338>

674 Ruel, S.M., Choubert, J.M., Esperanza, M., Miège, C., Madrigal, P.N., Budzinski, H., Le Ménach,
675 K., Lazarova, V., Coquery, M., 2011. On-site evaluation of the removal of 100 micro-
676 pollutants through advanced wastewater treatment processes for reuse applications. *Water*
677 *Sci. Technol.* 63, 2486. <https://doi.org/10.2166/wst.2011.470>

678 Schollée, J.E., Bourgin, M., von Gunten, U., McArdell, C.S., Hollender, J., 2018. Non-target
679 screening to trace ozonation transformation products in a wastewater treatment train
680 including different post-treatments. *Water Res.* 142, 267–278.
681 <https://doi.org/10.1016/j.watres.2018.05.045>

682 Sgroi, M., Anumol, T., Roccaro, P., Vagliasindi, F.G.A., Snyder, S.A., 2018. Modeling emerging
683 contaminants breakthrough in packed bed adsorption columns by UV absorbance and
684 fluorescing components of dissolved organic matter. *Water Res.*
685 <https://doi.org/10.1016/j.watres.2018.09.018>

686 Sgroi, M., Roccaro, P., Korshin, G.V., Vagliasindi, F.G.A., 2017. Monitoring the Behavior of
687 Emerging Contaminants in Wastewater-Impacted Rivers Based on the Use of
688 Fluorescence Excitation Emission Matrixes (EEM). *Environ. Sci. Technol.* 51, 4306–
689 4316. <https://doi.org/10.1021/acs.est.6b05785>

690 Sgroi, M., Roccaro, P., Oelker, G., Snyder, S.A., 2016. N-nitrosodimethylamine (NDMA)
691 formation during ozonation of wastewater and water treatment polymers. *Chemosphere*
692 144, 1618–1623. <https://doi.org/10.1016/j.chemosphere.2015.10.023>

693 Völker, J., Stapf, M., Miehe, U., Wagner, M., 2019. Systematic Review of Toxicity Removal by
694 Advanced Wastewater Treatment Technologies via Ozonation and Activated Carbon.
695 *Environ. Sci. Technol.* 53, 7215–7233. <https://doi.org/10.1021/acs.est.9b00570>

696 von Gunten, U., 2003a. Ozonation of drinking water: Part I. Oxidation kinetics and product
697 formation. *Water Res.* 37, 1443–1467. [https://doi.org/10.1016/S0043-1354\(02\)00457-8](https://doi.org/10.1016/S0043-1354(02)00457-8)

698 von Gunten, U., 2003b. Ozonation of drinking water: Part II. Disinfection and by-product
699 formation in presence of bromide, iodide or chlorine. *Water Res.* 37, 1469–1487.
700 [https://doi.org/10.1016/S0043-1354\(02\)00458-X](https://doi.org/10.1016/S0043-1354(02)00458-X)

701 Vulliet, E., Cren-Olivé, C., Grenier-Loustalot, M.-F., 2011. Occurrence of pharmaceuticals and
702 hormones in drinking water treated from surface waters. *Environ. Chem. Lett.* 9, 103–114.
703 <https://doi.org/10.1007/s10311-009-0253-7>

704 Zhang, S., Wu, C., Zhou, Y., Wang, Y., He, X., 2018. Effect of wastewater particles on catalytic
705 ozonation in the advanced treatment of petrochemical secondary effluent. *Chem. Eng. J.*
706 345, 280–289. <https://doi.org/10.1016/j.cej.2018.03.184>

707 Zietzschmann, F., Mitchell, R.-L., Jekel, M., 2015. Impacts of ozonation on the competition
708 between organic micro-pollutants and effluent organic matter in powdered activated
709 carbon adsorption. *Water Res.* 84, 153–160. <https://doi.org/10.1016/j.watres.2015.07.031>

710 Zietzschmann, F., Stützer, C., Jekel, M., 2016. Granular activated carbon adsorption of organic
711 micro-pollutants in drinking water and treated wastewater – Aligning breakthrough curves
712 and capacities. *Water Res.* 92, 180–187. <https://doi.org/10.1016/j.watres.2016.01.056>

713 Zietzschmann, F., Worch, E., Altmann, J., Ruhl, A.S., Sperlich, A., Meinel, F., Jekel, M., 2014.
714 Impact of EfOM size on competition in activated carbon adsorption of organic micro-
715 pollutants from treated wastewater. *Water Res.* 65, 297–306.
716 <https://doi.org/10.1016/j.watres.2014.07.043>

717 Zimmermann, S.G., Wittenwiler, M., Hollender, J., Krauss, M., Ort, C., Siegrist, H., von Gunten,
718 U., 2011. Kinetic assessment and modeling of an ozonation step for full-scale municipal
719 wastewater treatment: Micropollutant oxidation, by-product formation and disinfection.
720 *Water Res.* 45, 605–617. <https://doi.org/10.1016/j.watres.2010.07.080>

721 Zucker, I., Lester, Y., Avisar, D., Hübner, U., Jekel, M., Weinberger, Y., Mamane, H., 2015.
722 Influence of Wastewater Particles on Ozone Degradation of Trace Organic Contaminants.
723 Environ. Sci. Technol. 49, 301–308. <https://doi.org/10.1021/es504314t>

724