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1 **Biogenic Fenton reaction – a possible mechanism for the mineralization of**  
2 **organic carbon in fresh waters**

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12 **ABSTRACT:** To explore the mechanisms that mineralize poorly bioavailable natural organic  
13 carbon (OC), we measured the mineralization of OC in two lake waters over long-term  
14 experiments (up to 623 days) at different pH and iron (Fe) levels. Both microbial and  
15 photochemical mineralization was higher at pH acidified to 4 than at the ambient pH 5 or an  
16 elevated pH 6. During 244 days, microbes mineralized up to 60% of OC in 10- $\mu$ m filtrates of  
17 lake water and more than 27% in 1- $\mu$ m filtrates indicating that large-sized microbes/grazers  
18 enhance the mineralization of OC. A reactivity continuum model indicated that acidification  
19 stimulated the microbial mineralization of OC especially in the later (>weeks) phases of  
20 experiment when the bioavailability of OC was poor. The reactive oxygen species produced by  
21 light or microbial metabolism could have contributed to the mineralization of poorly  
22 bioavailable OC through photochemical and biogenic Fenton processes catalyzed by  
23 indigenous Fe in lake water. When Fe was introduced to artificial lake water to the  
24 concentration found in the study lakes, it increased the densities of bacteria growing on solid  
25 phase extracted dissolved organic matter and in a larger extent at low pH 4 than at pH 5. Our  
26 results suggest that in addition to the photochemical Fenton process (photo-Fenton), microbes  
27 can transfer poorly bioavailable OC into labile forms and CO<sub>2</sub> through extracellular Fe-  
28 catalyzed reactions (i.e., biogenic Fenton process).

29

30 *Keywords:*

31 Organic carbon

32 Reactive oxygen species

33 Iron

34 Biogenic Fenton

35 Microbes

36 Reactivity continuum

## 37 1. Introduction

38 In fresh waters, the mineralization of natural organic carbon (OC) emits 2.1 Pg CO<sub>2</sub>-C  
39 yr<sup>-1</sup> to the atmosphere (Raymond et al., 2013). Solar radiation-induced photochemical reactions  
40 can account for one tenth of the CO<sub>2</sub> emission (Aarnos et al., 2018; Koehler et al., 2014).  
41 Additional mechanisms are needed for the mineralization of OC in fresh waters with a typical  
42 first order decay coefficient of ~0.00076 d<sup>-1</sup> corresponding to approximately 2.5 years half-  
43 lives (Catalán et al., 2016). The mechanisms responsible for the mineralization of poorly  
44 bioavailable OC are mostly unknown and have been seldom addressed with long term  
45 experiments (Koehler et al., 2012).

46 Reactive oxygen species (ROS) may contribute to the slow mineralization of poorly  
47 bioavailable OC (Waggoner et al., 2017). Three major processes produce ROS in the  
48 environment. (i) ROS are produced at redoxclines when reduced forms of dissolved organic  
49 matter (DOM), iron (Fe) or other metals enter from anoxic to oxic strata and react with O<sub>2</sub>  
50 (Liao et al., 2019; Minella et al., 2015; Page et al., 2012, 2013; Trusiak et al., 2018; Waggoner  
51 et al., 2017). (ii) Photochemistry produces ROS at narrow surface strata during daytime  
52 (Micinski et al., 1993; Wolf et al., 2018; Zepp et al., 1992). (iii) Microbes produce ROS over  
53 the entire oxic water column (Diaz & Plummer, 2018; Dixon et al., 2013; Zhang et al., 2016).  
54 When integrated over the water column and the 24 hours of day, the production rate of  
55 superoxide (O<sub>2</sub><sup>•-</sup>) and hydrogen peroxide (H<sub>2</sub>O<sub>2</sub>) has been an order of magnitude larger through  
56 biology than photochemistry (Vermilyea et al., 2010). Thus, microbes have a high potential to  
57 mineralize OC through ROS reactions. O<sub>2</sub><sup>•-</sup> and H<sub>2</sub>O<sub>2</sub> mineralize OC poorly, but they can be  
58 reduced to hydroxyl radical (•OH) through the Fenton process (Gligorovski et al., 2015). In  
59 fresh waters, •OH reacts primarily with DOC and can transform it through organic  
60 intermediates into CO<sub>2</sub> (Goldstone et al., 2002; Vione et al., 2014).

61

62 Photochemical and biogenic Fenton processes generate  $\bullet\text{OH}$ . Solar radiation-induced  
63 photochemical production of the Fenton reactants, Fe(II) and  $\text{H}_2\text{O}_2$ , initiates the photochemical  
64 Fenton process (photo-Fenton; Faust & Zepp, 1993; Vione et al., 2014; Voelker et al., 1997).  
65 Bacteria can degrade OC through biogenic Fenton process (bio-Fenton) (Gu et al., 2016, 2018;  
66 Ma et al., 2016; Sekar & DiChristina, 2014; Xiao et al., 2016). For example, bacteria reduce  
67 Fe(III) to Fe(II) and the oxidation of Fe(II) generates  $\bullet\text{OH}$  (Sekar & DiChristina, 2014).  
68 Alternatively, bacteria produce extracellular  $\text{O}_2^{\bullet-}$ , a precursor for  $\text{H}_2\text{O}_2$  and siderophores that  
69 reduce Fe(III) to Fe(II) (Gu et al., 2018). In this study, the bio-Fenton process refers to the  
70 metabolic pathways that lead to extracellular Fe(II) and  $\text{H}_2\text{O}_2$  followed by abiotic Fenton  
71 process that produces  $\bullet\text{OH}$ .

72 Hydroxyl radicals have short life times ( $\sim\mu\text{s}$ ), extremely low concentrations ( $\leq 10^{-15}$ – $10^{-18}$   
73  $\text{mol L}^{-1}$ ) and their detection is difficult at the time scale of OC turnover (Burns et al.,  
74 2012). Therefore alternative approaches are needed to evaluate a possible role of  $\bullet\text{OH}$  on the  
75 long-term mineralization of poorly bioavailable OC. For example, long-term experiments can  
76 be designed either to favor or hinder the production of  $\bullet\text{OH}$  radicals. The rate of  $\bullet\text{OH}$  radical  
77 production by the Fenton process is negligible in the absence of dissolved Fe (which would  
78 function as a catalyst), but increases with the concentration of Fe (Christoforidis et al., 2015;  
79 Rush & Bielski, 2005). Although the Fenton process produces  $\bullet\text{OH}$  radicals at neutral pH, low  
80 pH increases the rates of  $\bullet\text{OH}$  production (Georgi et al., 2007; Pignatello et al., 2006; Zepp et  
81 al., 1992). The rates of bio-Fenton should increase with the increasing size of microbial  
82 community, because the number of ROS producers increases and may include eukaryotic  
83 microbes (Diaz & Plummer, 2018), which possibly explain why  $>5\text{-}\mu\text{m}$  size fraction were  
84 responsible for  $>85\%$  of ROS production in an earlier study (Zhang et al., 2016).

85 We hypothesize that the bio-Fenton process along with the photo-Fenton contribute to  
86 the long-term mineralization of poorly bioavailable OC. The hypothesis was tested with long-

87 term (up to 623 days) experiments that measured microbial and photochemical mineralization  
88 of OC in 1- $\mu\text{m}$  and 10- $\mu\text{m}$  filtrates of two lake waters, and assessed bacterial growth on solid  
89 phase extracted dissolved organic matter (SPE-DOM) at different Fe and pH levels. Gamma  
90 reactivity continuum model (Vähätalo et al., 2010; Arndt et al. 2013) described the rate  
91 constants for the microbial mineralization of OC separately in the early and the late phases of  
92 bioassays corresponding to the labile and poorly bioavailable fractions of OC, respectively. If  
93 the latter and the total amount of mineralized OC associates positively with experimental  
94 acidification, the concentration of Fe, and large-sized microbial community, the associations  
95 indicate the mineralization of poorly bioavailable OC through the bio-Fenton process. If  
96 acidification enhances photochemical mineralization, it indicates that part of OC is mineralized  
97 through the photo-Fenton process.

## 98 **2. Materials and methods**

### 99 *2.1 Water sampling, DOM extraction, DOM-Fe, artificial lake water and microbial isolate*

100 Surface water samples (0–1 m) were collected between July and October from Lake  
101 Vakea-Kotinen and Iso Valkjärvi in Finland (Table 1, lake characteristics given in Table S1  
102 and Text S-III). For the experiment with different levels of introduced Fe(III), the SPE-  
103 extractable part of DOM (typically >60% of total DOC) was isolated from Lake Valkea-  
104 Kotinen (“Fe” experiment, Table 1). The SPE followed the method of Dittmar et al. (2008)  
105 except we introduced 0.01 M sodium fluoride (NaF, Sigma-Aldrich) to filtered (<0.2  $\mu\text{m}$ ) lake  
106 water to exchange Fe(III) from DOM to fluoride ligands. SPE removed 96.6% of Fe from lake  
107 water but the SPE-DOM nevertheless contained 8.5 nmol Fe per milligram DOM to satisfy the  
108 microbial requirement of Fe in the “Fe” experiment.

109 DOM-Fe(III) complexes were prepared from  $\text{FeCl}_3 \cdot 6\text{H}_2\text{O}$  (Sigma-Aldrich) and SPE-  
110 DOM. Acidified (pH 2, HCl) SPE-DOM solution (50 mg  $\text{L}^{-1}$  in ultrapure water) received 1  
111 mM Fe(III) and was titrated with NaOH to pH 4 or 5. During the titration, the binding sites of

112 DOM suppressed the hydrolysis of Fe(III) and DOM-Fe was formed (Karlsson & Persson,  
113 2012). Finally, the solution of DOM-Fe received a stock solution of inorganic ions (Table S2)  
114 to simulate the composition of lake water in Valkea-Kotinen (“Fe” experiment, Table 1).

115 A grazer-free microbial community for “Fe” experiment was isolated from the same  
116 water sample as SPE-DOM as described in Xiao et al. (2016).

## 117 *2.2 Experimental procedures*

### 118 *2.2.1 Microbial mineralization of OC – microbial size-fractions and different pH levels*

119 To examine the mineralization of OC by small- or large-size microbes, lake waters were  
120 filtered either through 1- $\mu$ m or 10- $\mu$ m (Nuclepore) filters, respectively (the experiments “1- $\mu$ m  
121 dark” and “1- $\mu$ m or 10- $\mu$ m dark”; Table 1). For assessing the effect of pH on the mineralization  
122 of OC, the ambient pH of filtrates (5.24–5.45) was adjusted with H<sub>2</sub>SO<sub>4</sub> or NaOH to pH 4, 5  
123 or 6 (Table 1). Eventually within a day of sample collection, 5 mL of the pH adjusted filtrates  
124 were sealed in pre-combusted (450°C for 2 h) clear borosilicate glass ampoules with an  
125 approximately 7.5 mL headspace of air (Text S-I; McDowell et al., 1987; Salonen & Kononen,  
126 1984) and incubated at room temperature (approximately 23°C) in the dark up to 584 days.  
127 Three ampoules were periodically sacrificed for the determination of inorganic carbon (IC)  
128 content to calculate the mineralization of OC along the incubation as described in 2.3.1.

### 129 *2.2.2 Photochemical mineralization of OC alone or together with microbes*

130 The ampoule experiments explained above were modified to assess the mineralization  
131 of OC through abiotic photochemistry alone or combined with microbial metabolism at pH 4  
132 or 5 (the experiments “0.1- $\mu$ m photochemistry” and “1- $\mu$ m light or dark”, respectively; Table  
133 1, details in Text S-I). For the “1- $\mu$ m light or dark”-experiment, lake water was filtered through  
134 1- $\mu$ m, adjusted to pH 5 or 4 and sealed in the ampoules. For the “0.1- $\mu$ m photochemistry”-  
135 experiment, the 1- $\mu$ m filtrates were filtrated further through 0.1- $\mu$ m, adjusted to pH 5 or 4,  
136 sealed in ampoules and autoclaved. Half of the ampoules received irradiance from fluorescent

137 lamps at 15°C (Figure S1) and the remaining half (the dark controls) were kept in the dark at  
138 the same temperature up to 623 days (Table 1).

### 139 *2.2.3 Microbial growth on DOC at different levels of Fe and pH*

140 “Fe” experiment examined the effect of both pH and Fe on microbial growth on DOC  
141 (“Fe” experiment in Table 1). DOM-Fe(III) (0, 5 or 20 µM Fe) was dissolved to artificial lake  
142 water (the final concentration of DOC, 9.8 mg L<sup>-1</sup>, Table S2) and adjusted to pH 4 or 5. The  
143 isolated grazer-free bacterial community from Lake Valkea Kotinen (Xiao et al. (2016) was  
144 inoculated (7% vol/vol) in the artificial lake waters with DOM-Fe(III) and incubated in the  
145 dark at 23°C for 28 days. During the incubation, microbial growth was periodically assessed  
146 as bacterial densities.

## 147 *2.3 Analytical methods*

### 148 *2.3.1 Mineralization of OC*

149 The content of IC in the ampoules was periodically measured as CO<sub>2</sub> after purging  
150 dissolved IC in lake water together with CO<sub>2</sub> in an air headspace to a carbon analyzer (Text S-  
151 I; Figure S2; Salonen, 1981). An increase in IC during the experiments described the amount  
152 of mineralized OC. The concentrations of OC along the experiment in Figure 1 were calculated  
153 by subtracting the mineralized OC from the concentration of OC determined from GF/C-  
154 filtered (nominal pore size 1.2 µm, Whatman) lake water prior to experiments by high-  
155 temperature combustion (Salonen, 1979).

### 156 *2.3.2 Bacterial densities*

157 Bacterial samples were periodically fixed (final concentration of 1% paraformaldehyde  
158 with 0.05% glutaraldehyde) from the Fe-experiment and counted with a BD FACSCalibur™  
159 flow cytometer (BD Biosciences, USA) using SYBR Green I (Sigma-Aldrich) as nucleic acid  
160 stain (Gasol & Del Giorgio, 2000).



### 161 2.3.3 Reactivity continuum modeling of OC mineralization

162 The mineralization of OC was described by the gamma reactivity continuum model,  
163 which expresses mathematically the conceptual decomposition processes that remove  
164 preferentially the most labile parts of OC and shift the reactivity continuum of OC toward poor  
165 bioavailability with time (Arndt et al., 2013; Catalán et al., 2016; Vähätalo et al., 2010):

$$166 \text{OC}(t) = \text{OC}(t_0) (a (a + t)^{-1})^v \quad (1),$$

167 where  $\text{OC}(t)$  is the concentration of OC ( $\text{mg C L}^{-1}$ ) at time  $t$  (d),  $\text{OC}(t_0)$  is the initial  
168 concentration of OC,  $a$  (d) and  $v$  (dimensionless) are fitting parameters (Koehler et al., 2012).  
169 The values of parameters were determined by the curve fitting toolbox version 3.5.2 of Matlab  
170 R2015b (The MathWorks Inc.) using non-linear least squares method and trust-region  
171 algorithm (Vähätalo et al., 2010).

172 The first order decay coefficient of OC at time  $t$ ,  $k(t)$ , was described as:

$$173 k(t) = v (a + t)^{-1} \quad (2).$$

174 At time  $t = 0$ ,  $k(t_0) = v/a$  ( $\text{d}^{-1}$ ) expresses the initial first order decay coefficient. Although the  
175 value of  $k(t)$  depends on both  $v$  and  $a$ , the value of  $a$  has the largest impact on the value of  $k(t)$   
176 in the early phase of mineralization process (Arndt et al., 2013). In the early phase of  
177 mineralization, the value of  $a$  describes the average lifetime of more reactive OC components  
178 and small values of  $a$  refer to high values of  $k(t)$  (Arndt et al., 2013). In the late phases of  
179 decomposition when the value of  $t \gg a$ , the value of  $k(t)$  depends mainly on  $v$  and high values  
180 of  $v$  refer to high values of  $k(t)$  for poorly bioavailable OC (Arndt et al., 2013).

### 181 2.4 Statistical analyses

182 All experiments had three replicates at each time point. The differences between  
183 treatments were tested with paired  $t$  test with two-tailed distributions.

### 184 3. Results

#### 185 3.1 Microbial mineralization of OC in 1- $\mu$ m filtered lake waters adjusted to different pHs

186 In the first experiment, we tested a hypothesis that the ambient concentration of Fe in  
187 our lake waters (3.2–4.5  $\mu$ M, Table S1) is sufficient to induce the biogenic Fenton process,  
188 which due to its pH dependence increases mineralization of OC at low pH. When 1- $\mu$ m filtered  
189 lake water with small-sized microbes was enclosed in ampoules and incubated in the dark at  
190 23°C, microbes mineralized up to  $26.9 \pm 0.4\%$  and  $24.2 \pm 0.3\%$  of OC in water from Lake  
191 Valkea-Kotinen and Lake Iso Valkjärvi, respectively, during 584 days (Figure 1a&b). The  
192 markers of Figure 1 show the experimental data and the curves illustrate the concentration of  
193 OC calculated according to the gamma model (Eq. 1) using the values of  $a$  and  $v$  reported in  
194 Table 2. An acidification of lake water to pH 4 increased the microbial mineralization of OC  
195 compared to the treatments adjusted at higher pH 5 or 6 ( $t$ -test,  $P < 0.05$ ,  $n = 3$ ; Figure 1a&b)  
196 and supported our hypothesis.

197 We further hypothesized that microbes mineralize first the labile OC through direct  
198 uptake or enzymatic hydrolysis independently of the bio-Fenton, but after the depletion of  
199 labile OC the contribution of bio-Fenton process to the mineralization of OC increases in the  
200 late phase of biodegradation. The gamma model can assess the impact of pH on the first order  
201 rate constants for mineralization of OC,  $k(t)$ , separately in the initial, the early and the late  
202 phase of biodegradation (Figure 2a&b, Table 2). In the 1- $\mu$ m filtrate of Lake Valkea-Kotinen,  
203 the initial  $k(t_0)$  was similar at pH 4 and 5 ( $k(t_0) = 0.0021$ – $0.0022$  d<sup>-1</sup>; Table 2). The value of  $a$   
204 was larger at pH 4 (64.5 d) than at pH 5 (38.1 d) indicating that the average lifetime of more  
205 reactive OC components was longer in the acidified treatment (Table 2). In both lake waters,  
206 the values of  $k(t)$  remained higher at pH 4 than at pH 5 or 6 in the late phase of experiment  
207 (Figure 2a&b). The values of  $v$  were higher at pH 4 than at 5 or 6 (Table 2) indicating that  
208 acidification promoted mineralization of OC in the late phase of biodegradation. Thus along

209 with our hypothesis, the acidification of lake waters to pH 4 did not necessary change the initial  
210 microbial mineralization rates and even slowed down the consumption of more reactive OC  
211 components in the early phase (high values of  $a$ ) but caused elevated rates of mineralization in  
212 late phase of biodegradation (high values of  $v$ ; Figure 2a&b, Table 2).

### 213 *3.2 Microbial mineralization of OC in 1- $\mu$ m and 10- $\mu$ m filtrates at different pHs*

214 Based on an earlier observation that large-sized microbes are primarily responsible for  
215 the production of ROS (Zhang et al., 2016) and we hypothesized higher rates of bio-Fenton  
216 reactions in 10- $\mu$ m than 1- $\mu$ m filtrates of lake water. When the microbial mineralization of OC  
217 in two size fractions is compared, microbes mineralized more OC in the 10- $\mu$ m than in the 1-  
218  $\mu$ m filtrates and more at pH 4 than at pH 5 (Figure 1c&d). For example, at the end of experiment  
219 (day 244) in the 10- $\mu$ m filtrate of Lake Valkea-Kotinen, microbes had mineralized  $60.1 \pm 3.0\%$   
220 of OC at pH 4 and more than  $26.6 \pm 0.8\%$  at pH 5, which is close to the ambient pH of lake  
221 water (Figure 1c).

222 The  $a$ -values were lower but the values of  $v$  typically were higher in the 10- $\mu$ m than in  
223 the 1- $\mu$ m filtrate within each pH-treatment (Table 2). These kinetic parameters indicate that in  
224 the early phase of biodegradation, the large-sized microbes consumed quickly the labile OC  
225 most likely without a notable contribution from the bio-Fenton process. In the late phases of  
226 experiment, high values of  $v$  and the extensive amount of mineralized OC indicate that large-  
227 sized microbes were able to mineralize poorly bioavailable OC extensively possibly through  
228 the bio-Fenton process because the acidification to pH 4 again increased the mineralization  
229 (Figure 2c&d, Table 2).

### 230 *3.3 Photochemical mineralization of lake water DOC at different pHs*

231 If the photo-Fenton process contributes to the photochemical mineralization of DOC,  
232 the mineralization of DOC should increase in irradiated acidified waters because low pH  
233 promotes the photo-Fenton process. Irradiation mineralized up to  $14.8 \pm 0.5\%$  and  $12.4 \pm 0.3\%$

234 of DOC in the autoclaved 0.1- $\mu\text{m}$  filtrates of Lake Valkea-Kotinen and Lake Iso Valkjärvi,  
235 respectively, by the end of the 623 d experiment (Figure 1e&f). In the dark controls, the  
236 mineralization of DOC remained negligible (Figure 1e&f). The amount of photochemically  
237 mineralized DOC and the values of  $k(t)$  for the photochemical mineralization were larger at pH  
238 4 than at pH 5 (Figure 2e&f; Table 3), which agrees with an elevated rate of photo-Fenton  
239 process at acidic conditions.

#### 240 *3.4 Combined photochemical and microbial mineralization of OC at different pHs*

241 When microbes were present in irradiated waters, the irradiation and lower pH  
242 increased the mineralization of OC (Figure 1g&h). In water from Lake Valkea-Kotinen, the  
243 irradiation stimulated the biological mineralization of OC, because the difference in the amount  
244 of mineralized OC between the irradiated and the dark control treatments was larger in the 1-  
245  $\mu\text{m}$  filtrates than in the autoclaved 0.1  $\mu\text{m}$  filtrates (Table 3).

246 The values of  $k(t)$  and  $v$  were larger at the low pH and in the irradiated 1- $\mu\text{m}$  filtrates  
247 than in the corresponding dark controls (Figure 2g&h, Table 2). The high values of  $v$  in the  
248 irradiated 1- $\mu\text{m}$  filtrates and at low pH indicate a contribution of Fenton process to the  
249 mineralization of poorly bioavailable OC in the late phases of experiment.

#### 250 *3.5 Effects of Fe on microbial growth on lake water DOC*

251 Based on the experiments with filtered lake waters (Figure 1), it is clear that low pH  
252 increased the mineralization of OC (Table 4) and in particular at the late phases of  
253 biodegradation (Figure 2; higher values of  $v$  at pH 4 than pH 5, Table 2). These observations  
254 suggest that the ambient concentration of Fe was high enough to support the Fenton process in  
255 our lake waters. The concentration of Fe was higher in Lake Valkea-Kotinen than in Lake Iso  
256 Valkjärvi (Table S1) and accordingly the values of  $v$  were typically higher for microbial  
257 mineralization of OC in the water from Lake Valkea-Kotinen than from Lake Iso Valkjärvi  
258 (Table 2). These results indicate that a higher concentration of Fe increased the contribution of

259 bio-Fenton process to the mineralization of poorly bioavailable DOC in the late phases of  
260 biodegradation.

261 To further study whether microbial growth on DOC depends on the concentration of Fe  
262 in addition to pH, we examined how microbes from Lake Valkea-Kotinen grow on SPE-DOM  
263 from Lake Valkea-Kotinen with or without introduced Fe at pH 4 or 5 (Figure 3). In the SPE-  
264 DOM dissolved in artificial lake water with 0.17  $\mu\text{M}$  Fe, microbes grew similarly at both pH  
265 levels (Figure 3a). When the SPE-DOM was complexed with 5  $\mu\text{M}$  Fe(III) approximating the  
266 ambient concentration of Fe in Lake Valkea Kotinen (Table S1), microbial growth increased  
267 compared to SPE-DOM without introduced Fe (compare Figure 3a and 3b). In the presence of  
268 5  $\mu\text{M}$  SPE-DOM-Fe, bacteria reached higher densities at pH 4 than at pH 5 during 28-day  
269 experiment (*t*-test,  $P < 0.05$ ,  $n = 3$ ; Figure 3b). Microbes reached highest densities when they  
270 grew on SPE-DOM with 20  $\mu\text{M}$  Fe (compare Figure 3c to 3a&b) and the final densities were  
271 higher at pH 4 than at pH 5 (*t*-test,  $P < 0.05$ ,  $n = 3$ ; Figure 3c). Thus, the combination of Fe and  
272 low pH increased the growth of microbes on DOC. The final density of microbes in the end of  
273 experiment increased with the concentration of Fe (compare the panels a, b and c in Figure 3)  
274 indicating that the bio-Fenton process supported higher bacterial density with increasing  
275 concentration of Fe.

#### 276 4. Discussion

277 Our experiment with SPE-DOM shows that Fe enhances microbial growth on DOC and  
278 in particular in acidic water (Figure 3). This study further shows that a decrease in pH from 5  
279 to 4 increases the biological mineralization of OC in lake waters containing the ambient  
280 concentration of Fe (Figure 1), but does not enhance the growth of microbes on SPE-DOM  
281 extract without introduced Fe (Figure 3a). Acidification increases also the photochemical  
282 mineralization of DOC in this and many earlier studies, but not in waters with low  
283 concentrations of Fe (Gu et al., 2017 and references therein). In this study, the combination of

284 low pH and Fe enhances both photochemical and biological mineralization of OC. A plausible  
285 explanation for the enhancement is the Fenton reaction either driven by microbial metabolism  
286 (bio-Fenton) or light (photo-Fenton).

#### 287 *4.1 Photochemical Fenton process*

288 The photo-Fenton process provides an explanation for an increase in the photochemical  
289 mineralization of DOC with decreasing pH observed in this and earlier studies (Gu et al., 2017  
290 and references therein). In the photo-Fenton process, irradiation generates the Fenton reactants  
291 through a series of reactions that start from the light absorption by CDOM or DOM-Fe(III)  
292 complexes illustrated as the processes [10] and [11], respectively (Figure 4). The ligand-to-  
293 metal charge transfer in DOM-Fe(III) complexes ([11] in Figure 4) can mineralize a part of  
294 DOC to CO<sub>2</sub> and produce Fe(II). The photochemical oxidation of CDOM can reduce O<sub>2</sub> to  
295 O<sub>2</sub><sup>•-</sup> ([10] and [2] in Figure 4). Superoxide may reduce Fe(III) to Fe(II) ([4] in Figure 4) or  
296 lead to the production of H<sub>2</sub>O<sub>2</sub> ([3] in Figure 4). Finally, the Fenton reaction ([5] in Figure 4)  
297 produces <sup>•</sup>OH that transforms DOC into labile forms or CO<sub>2</sub> ([6] in Figure 4). The microbial  
298 consumption of labile forms can explain the enhanced microbial mineralization of OC in the  
299 irradiated waters of present study ([7] in Figure 4, Table 3).

300 In the present and many earlier studies, acidity increases the photochemical  
301 mineralization of DOC and this pH dependence associates with the photo-Fenton process (Gu  
302 et al., 2017 and references therein). Low pH promotes (i) the protonation of O<sub>2</sub><sup>•-</sup> to its  
303 conjugate acid (HO<sub>2</sub><sup>•</sup> [2] in Figure 4), (ii) the dismutation of O<sub>2</sub><sup>•-</sup>/HO<sub>2</sub><sup>•</sup> to H<sub>2</sub>O<sub>2</sub> and (iii) the  
304 turnover of Fe(II)-Fe(III) ([3] and [11] in Figure 4; Garg et al., 2015; Rush & Bielski, 1985).  
305 The turnover of Fe(II)-Fe(III) is high at low pH, because acidity increases the binding of Fe(III)  
306 on DOM into soluble reactive forms ([9] in Figure 4; Neubauer et al., 2013). Additionally, the  
307 Fenton process produces <sup>•</sup>OH at low pH ([5] in Figure 4), but ferryl iron (Fe(IV) or Fe(V))  
308 with a lower oxidation capacity at higher pH (Vione et al., 2014). According to an earlier study

309 with different pH and Fe levels, an acidification from pH 5 to pH 4 increases the photochemical  
310 mineralization of DOC by 32% in 10 mg L<sup>-1</sup> DOM associated with 3 μM Fe approximating the  
311 conditions in the lakes of present study (Table 4; Gu et al., 2017). The corresponding acidity-  
312 induced increase in photochemical mineralization in this study is similar (27–35%) to the  
313 earlier estimate (32%; Table 4; Gu et al. 2017). In the absence of Fe, acidification does not  
314 change the rate of photochemical mineralization (Gu et al., 2017) indicating that the impact of  
315 acidification found in this study is connected to the photo-Fenton process catalyzed by the  
316 ambient concentrations of Fe in the examined lake waters.

#### 317 4.2 Biogenic Fenton process

318 If photochemically produced ROS can initiate the photo-Fenton process in our lake  
319 waters, ROS produced by microbes should be able to initiate the bio-Fenton process. The bio-  
320 Fenton process can provide a mechanistic explanation for the enhanced microbial  
321 mineralization of poorly bioavailable OC in the late phases of biodegradation at low pH and  
322 the enhanced growth of bacteria on our SPE-DOM-Fe(III) with increasing concentration of Fe.  
323 In this study ~~and~~ in oxic surface waters with DOM-Fe(III) in general, a plausible start for the  
324 bio-Fenton process is a transport of electron from the cellular metabolism to O<sub>2</sub> for the  
325 production of O<sub>2</sub><sup>•-</sup> ([1–2] in Figure 4; Diaz et al., 2013). The produced O<sub>2</sub><sup>•-</sup> can initiate a series  
326 of abiotic reactions that eventually lead to the Fenton reaction. Several mechanisms can reduce  
327 O<sub>2</sub><sup>•-</sup> to H<sub>2</sub>O<sub>2</sub> ([3] in Figure 4; Garg et al., 2011; Petasne & Zika, 1987) and O<sub>2</sub><sup>•-</sup> can reduce  
328 Fe(III) bound on DOM to Fe(II) ([4] in Figure 4; Halliwell, 1978; Yuan et al., 2016). H<sub>2</sub>O<sub>2</sub> and  
329 Fe(II) undergo the Fenton reaction and produce •OH, which breaks down OC into CO<sub>2</sub> and  
330 labile forms ([5]–[6] in Figure 4; Goldstone et al., 2002; Zazo et al., 2005). Biology gets  
331 involved again when microbes take up labile OC, respire it to CO<sub>2</sub> and produce reducing  
332 equivalents (e.g., NADH) for oxidoreductases that generate extracellular O<sub>2</sub><sup>•-</sup> ([7]–[8], [1] in  
333 Figure 4). An introduction of O<sub>2</sub><sup>•-</sup> to the same 20 μM SPE-DOM-Fe(III) from Lake Valkea-

334 Kotinen dissolved in the same artificial lake water used in this study produces  $\bullet\text{OH}$  in  
335 autocatalytic manner and breaks down DOM (Xiao et al., 2020). The earlier study provides  
336 further evidence for the proposed bio-Fenton process, where microbially produced  $\text{O}_2^{\bullet-}$  reacts  
337 with DOM-Fe(III) and generates  $\bullet\text{OH}$ , which eventually breaks down OC.

#### 338 *4.3 Microbial size fractions*

339 In this study, microbes mineralize up to 5 times more OC in the 10- $\mu\text{m}$  than in the 1-  
340  $\mu\text{m}$  filtrates and the mineralization rates remain high up to 244 days, thus concerning also  
341 poorly bioavailable OC (Figure 1c&d). We attribute the elevated mineralization rates of poorly  
342 bioavailable OC to a more extensive production of ROS and bio-Fenton process in the 10- $\mu\text{m}$   
343 than 1- $\mu\text{m}$  filtrate (more details in Text S-IV). In an earlier study, the biological production of  
344  $\text{O}_2^{\bullet-}$  and  $\text{H}_2\text{O}_2$  decreased remarkably when water was filtered through a 5- $\mu\text{m}$  filter (Zhang et  
345 al., 2016). This finding together with the present study suggests that large-sized microbes in  
346 particular contribute to reactive species for degradation of recalcitrant OC.

#### 347 *4.4 Bio-Fenton process as a possible adaptation for the utilization of poorly bioavailable OC*

348 In this study, an acidification of lake water increases the microbial mineralization of  
349 OC in particular in the late phases of microbial succession (Figure 2; high values of  $v$  in Table  
350 2) and enhances bacterial growth on DOM-Fe(III) but only after a two–three weeks lag period  
351 (Figure 3). A similar lag time took place for bacteria growing on DOM-Fe(III) in an earlier  
352 study and involved drastic changes in the composition of bacterial community (Xiao et al.,  
353 2016). Our results and those of Xiao et al. (2016) suggest that microbes can adapt to the  
354 depletion of labile OC by promoting the bio-Fenton process to break down poorly bioavailable  
355 OC to labile forms that support microbial growth and mineralize OC.



356 *4.5 Mineralization of OC through the bio-Fenton process estimated from the dark production*  
357 *of H<sub>2</sub>O<sub>2</sub> and O<sub>2</sub><sup>•-</sup> in fresh waters*

358 Here we calculate how much the bio-Fenton process potentially contributed to the  
359 mineralization of OC in our experiments based on the dark production rates of ROS reported  
360 in the literature. According to the literature (Dixon et al., 2013; Marsico et al., 2015; Vermilyea  
361 et al., 2010; Zhang et al., 2016; Table S3 in SI), the dark production rates of extracellular O<sub>2</sub><sup>•-</sup>  
362 and H<sub>2</sub>O<sub>2</sub> correspond to an average of 4.0 μM e<sup>-</sup> d<sup>-1</sup> (range 0.7–15.4 μM e<sup>-</sup> d<sup>-1</sup>) when expressed  
363 as electrons transported from cytoplasm to extracellular milieu. According to a stoichiometry  
364 of •OH/ 2e<sup>-</sup> (Eq. S32 in Text S-VI of SI), this rate translates to 2.0 μM •OH d<sup>-1</sup> (range 0.4–7.7  
365 μM •OH d<sup>-1</sup>) or cumulatively to 1,200 μM •OH (range 240–4,620 μM •OH) during 20 months  
366 corresponding to a typical length of our experiments. If two •OHs mineralize OC to CO<sub>2</sub> (Eq.  
367 S35 in Text S-VII of SI), hydroxyl radicals mineralize cumulatively 600 μM (range 120–2,310  
368 μM) OC in 20 months, which is more than the observed microbial mineralization in our  
369 experiments (128 μM in 20 months; range 66–593 μM in 20 months; Figure 1). Superoxide  
370 dismutase and catalase enzymes as well as other sinks scavenge O<sub>2</sub><sup>•-</sup> and H<sub>2</sub>O<sub>2</sub>, and decrease  
371 the yield of •OH production per produced O<sub>2</sub><sup>•-</sup> or H<sub>2</sub>O<sub>2</sub> (Bielski et al., 1985). The yields for  
372 •OH production have been 1.4%–33% from the stoichiometry of •OH/2 e<sup>-</sup> (Page et al., 2012,  
373 2013). According to these yields, the bio-Fenton process mineralized 8.4–198 μM in our  
374 experiments during 20 months assuming a daily production of 4.0 μM extracellular e<sup>-</sup>. The  
375 calculations above suggest that the bio-Fenton process was potentially able to explain a  
376 remarkable fraction of OC mineralized in our long-term experiments.

377 Here we continue to calculate how much the bio-Fenton process can mineralize DOC  
378 in generic fresh water with a typical concentration of DOC (approximately 500 μM) and with  
379 a typical DOC half-life of approximately 2.5 years (Catalán et al., 2016; Text S-VIII in SI).

380 When accounting for the range of yields in  $\bullet\text{OH}$  production reported earlier (1.4%–33% from  
381  $\bullet\text{OH}/2\text{ e}^-$ ; Page et al., 2012, 2013), the mean production of  $4.0\ \mu\text{M e}^- \text{d}^{-1}$  corresponds to  
382  $0.027\text{--}0.64\ \mu\text{M } \bullet\text{OH d}^{-1}$  that can mineralize  $0.014\text{--}0.32\ \mu\text{M DOC d}^{-1}$ . As a non-selective  
383 oxidant  $\bullet\text{OH}$  likely mineralizes the poorly bioavailable rather than labile DOC because the  
384 poorly bioavailable fraction dominates the composition of DOC. The calculated daily rates are  
385 beyond the precision of conventional analytical techniques (e.g., for DOC) and masked by the  
386 fast turnover of labile DOC (e.g., in the respiration measurements). Therefore in the present  
387 study, the slow mineralization of poorly bioavailable became detectable only after the depletion  
388 of labile OC in the late phases of biodegradation or with a high precision technique (a bacterial  
389 density) under circumstances (high DOM-Fe(III) + acidity) that promoted bio-Fenton process.  
390 During the typical half-life of freshwater DOC, the bio-Fenton process however can mineralize  
391  $13\text{--}292\ \mu\text{M DOC}$  and account for 5.2–117% for the typical amounts of DOC (approximately  
392  $250\ \mu\text{M}$ ) mineralized in 2.5 years. These calculations indicate that the biogenic Fenton process  
393 can remarkably contribute to the turnover of DOC in fresh waters, but the large uncertainties  
394 in the calculation call upon further research.

## 395 **5. Conclusions**

396 The biogenic Fenton process couples the biogenic production of extracellular Fe(II) and  
397  $\text{H}_2\text{O}_2$  to the abiotic Fenton reaction that produces hydroxyl radicals. In oxic surface waters, the  
398 ubiquitous microbial extracellular production of superoxide can translate to  $\text{H}_2\text{O}_2$  and reduce  
399 DOM-Fe(III) to Fe(II). The subsequent Fenton reaction produces hydroxyl radicals that  
400 transform the poorly bioavailable OC into labile forms and  $\text{CO}_2$  at low rates. These rates are  
401 too low to be detected with short-term measurement but high enough to remarkably contribute  
402 to the turnover of OC in fresh waters.

403

404 **Author contributions**

405 A.V.V. and K.S. contributed to the design of the ampoule experiments. A.V.V. contributed to  
406 the preparation, sample collection and measurements of ampoule experiments. A.V.V. and Y.X.  
407 contributed to the design of Fe experiment. Y.X. contributed the preparation, sample collection  
408 and measurements of Fe experiment. All authors contributed to the writing and editing the  
409 manuscript.

410

411 **Declaration of competing interest**

412 The authors declare that there is no known competing financial interests or personal  
413 relationships that could have appear to influence the work reported in this article.

414

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420

421 **Appendix A. Supplementary information**

422 Supplementary information to this article can be found online at xxx.

423

424 **References**

425 Aarnos, H., Gélinas, Y., Kasurinen, V., Gu, Y., Puupponen, V.-M., & Vähätalo, A. V. (2018).  
426 Photochemical mineralization of terrigenous DOC to dissolved inorganic carbon in  
427 ocean. *Global Biogeochemical Cycles*, 32(2), 250–266.  
428 <https://doi.org/10.1002/2017GB005698>

429 Arndt, S., Jørgensen, B. B., LaRowe, D. E., Middelburg, J. J., Pancost, R. D., & Regnier, P.  
430 (2013). Quantifying the degradation of organic matter in marine sediments: A review  
431 and synthesis. *Earth-Science Reviews*. Elsevier.  
432 <https://doi.org/10.1016/j.earscirev.2013.02.008>

433 Bielski, B. H. J., Cabelli, D. E., & Arudi, R. L. (1985). Reactivity of HO<sub>2</sub>/O<sub>2</sub><sup>-</sup> radicals in  
434 aqueous solution. *Journal of Physical Chemistry Reference Data*, 14(4), 1041–1100.

435 Burns, J. M., Cooper, W. J., Ferry, J. L., King, D. W., DiMento, B. P., McNeill, K., et al.  
436 (2012). Methods for reactive oxygen species (ROS) detection in aqueous environments.  
437 *Aquatic Sciences*, 74(4), 683–734. <https://doi.org/10.1007/s00027-012-0251-x>

438 Catalán, N., Marcé, R., Kothawala, D. N., & Tranvik, L. J. (2016). Organic carbon  
439 decomposition rates controlled by water retention time across inland waters. *Nature*  
440 *Geoscience*, 9(7), 501–504. <https://doi.org/10.1038/ngeo2720>

441 Christoforidis, K. C., Louloudi, M., & Deligiannakis, Y. (2015). Effect of humic acid on  
442 chemical oxidation of organic pollutants by iron(II) and H<sub>2</sub>O<sub>2</sub>: A dual mechanism.  
443 *Journal of Environmental Chemical Engineering*, 3(4), 2991–2996.  
444 <https://doi.org/10.1016/j.jece.2015.02.005>

445 Cole, J. J., Prairie, Y. T., Kortelainen, P., Sobek, S., Tranvik, L. J., Prairie, Y. T., et al.  
446 (2007). Patterns and regulation of dissolved organic carbon: An analysis of 7,500 widely  
447 distributed lakes. *Limnology and Oceanography*, 52(3), 1208–1219.  
448 <https://doi.org/10.4319/lo.2007.52.3.1208>

449 Diaz, J. M., & Plummer, S. (2018). Production of extracellular reactive oxygen species by  
450 phytoplankton: past and future directions. *Journal of Plankton Research*, 40, 655–666.  
451 <https://doi.org/10.1093/plankt/fby039>

452 Diaz, J. M., Hansel, C. M., Voelker, B. M., Mendes, C. M., Andeer, P. F., & Zhang, T.  
453 (2013). Widespread production of extracellular superoxide by heterotrophic bacteria.  
454 *Science*, 340(6137), 1223–1226. <https://doi.org/10.1126/science.1237331>

455 Dittmar, T., Koch, B., Hertkorn, N., & Kattner, G. (2008). A simple and efficient method for  
456 the solid-phase extraction of dissolved organic matter (SPE-DOM) from seawater.  
457 *Limnology and Oceanography: Methods*, 6(6), 230–235.  
458 <https://doi.org/10.4319/lom.2008.6.230>

459 Dixon, T. C., Vermilyea, A. W., Scott, D. T., & Voelker, B. M. (2013). Hydrogen peroxide  
460 dynamics in an agricultural headwater stream: Evidence for significant  
461 nonphotochemical production. *Limnology and Oceanography*, 58(6), 2133–2144.  
462 <https://doi.org/10.4319/lo.2013.58.6.2133>

463 Faust, B. C., & Zepp, R. G. (1993). Photochemistry of aqueous Iron(III)-Polycarboxylate  
464 complexes: Roles in the chemistry of atmospheric and surface waters. *Environmental*  
465 *Science & Technology*, 27(12), 2517–2522. <https://doi.org/10.1021/es00048a032>

466 Garg, S., Rose, A. L., & Waite, T. D. (2011). Photochemical production of superoxide and  
467 hydrogen peroxide from natural organic matter. *Geochimica et Cosmochimica Acta*,  
468 75(15), 4310–4320. <https://doi.org/10.1016/j.gca.2011.05.014>

469 Garg, S., Jiang, C., & David Waite, T. (2015). Mechanistic insights into iron redox  
470 transformations in the presence of natural organic matter: Impact of pH and light.  
471 *Geochimica et Cosmochimica Acta*, 165, 14–34.  
472 <https://doi.org/10.1016/j.gca.2015.05.010>

473 Gasol, J. M., & Del Giorgio, P. A. (2000). Using flow cytometry for counting natural  
474 planktonic bacteria and understanding the structure of planktonic bacterial communities.  
475 *Scientia Marina*, 64(2), 197–224. <https://doi.org/10.3989/scimar.2000.64n2197>

476 Georgi, A., Schierz, A., Trommler, U., Horwitz, C. P., Collins, T. J., & Kopinke, F. D.  
477 (2007). Humic acid modified Fenton reagent for enhancement of the working pH range.  
478 *Applied Catalysis B: Environmental*, 72(1–2), 26–36.  
479 <https://doi.org/10.1016/j.apcatb.2006.10.009>

480 Gligorovski, S., Strekowski, R., Barbati, S., & Vione, D. (2015). Environmental implications  
481 of hydroxyl radicals ( $\bullet$ OH). *Chemical Reviews*, 115(24), 13051–13092.  
482 <https://doi.org/10.1021/cr500310b>

483 Goldstone, J. V., Pullin, M. J., Bertilsson, S., Voelker, B. M., & Hole, W. (2002). Reactions  
484 of hydroxyl radical with humic substances : Bleaching , mineralization , and production  
485 of bioavailable carbon substrates. *Environmental Science & Technology*, 36(3), 364–  
486 372. <https://doi.org/10.1021/ES0109646>

487 Gu, C., Wang, J., Liu, S., Liu, G., Lu, H., & Jin, R. (2016). Biogenic Fenton-like reaction  
488 involvement in cometabolic degradation of Tetrabromobisphenol A by *Pseudomonas* sp.  
489 *fz. Environmental Science & Technology*, 50(18), 9981–9989.  
490 <https://doi.org/10.1021/acs.est.6b02116>

491 Gu, C., Wang, J., Guo, M., Sui, M., Lu, H., & Liu, G. (2018). Extracellular degradation of  
492 tetrabromobisphenol A via biogenic reactive oxygen species by a marine  
493 *Pseudoalteromonas* sp. *Water Research*, 142, 354–362.  
494 <https://doi.org/10.1016/j.watres.2018.06.012>

495 Gu, Y., Lensu, A., Perämäki, S., Ojala, A., & Vähätalo, A. V. (2017). Iron and pH regulating  
496 the photochemical mineralization of dissolved organic carbon. *ACS Omega*, 2(5), 1905–  
497 1914. <https://doi.org/10.1021/acsomega.7b00453>

498 Halliwell, B. (1978). Superoxide-dependent formation of hydroxyl radicals in the presence of  
499 iron chelates. *FEBS Letters*, 92(2), 321–326. [https://doi.org/10.1016/0014-](https://doi.org/10.1016/0014-5793(78)80779-0)  
500 [5793\(78\)80779-0](https://doi.org/10.1016/0014-5793(78)80779-0)

501 Karlsson, T., & Persson, P. (2012). Complexes with aquatic organic matter suppress  
502 hydrolysis and precipitation of Fe(III). *Chemical Geology*, 322–323, 19–27.  
503 <https://doi.org/10.1016/j.chemgeo.2012.06.003>

504 Koehler, B., von Wachenfeldt, E., Kothawala, D., & Tranvik, L. J. (2012). Reactivity  
505 continuum of dissolved organic carbon decomposition in lake water. *Journal of*  
506 *Geophysical Research*, 117(G1), G01024. <https://doi.org/10.1029/2011JG001793>

507 Koehler, B., Landelius, T., Weyhenmeyer, G. A., Machida, N., & Tranvik, L. J. (2014).  
508 Sunlight-induced carbon dioxide emissions from inland waters. *Global Biogeochemical*  
509 *Cycles*, 28(7), 696–711. <https://doi.org/10.1002/2014GB004850>

510 Liao, P., Yu, K., Lu, Y., Wang, P., Liang, Y., & Shi, Z. (2019). Extensive dark production of  
511 hydroxyl radicals from oxygenation of polluted river sediments. *Chemical Engineering*  
512 *Journal*, 368, 700–709. <https://doi.org/10.1016/j.cej.2019.03.018>

513 Ma, J., Zhang, K., Huang, M., Hector, S. B., Liu, B., Tong, C., et al. (2016). Involvement of  
514 Fenton chemistry in rice straw degradation by the lignocellulolytic bacterium *Pantoea*  
515 *ananatis* Sd-1. *Biotechnology for Biofuels*, 9, 211. [https://doi.org/10.1186/s13068-016-](https://doi.org/10.1186/s13068-016-0623-x)  
516 [0623-x](https://doi.org/10.1186/s13068-016-0623-x)

517 Marsico, R. M., Schneider, R. J., Voelker, B. M., Zhang, T., Diaz, J. M., Hansel, C. M., &  
518 Ushijima, S. (2015). Spatial and temporal variability of widespread dark production and  
519 decay of hydrogen peroxide in freshwater. *Aquatic Sciences*, 77(4), 523–533.  
520 <https://doi.org/10.1007/s00027-015-0399-2>

521 McDowell, W., Cole, J., & Driscoll, C. (1987). Simplified version of the ampoule-persulfate  
522 method for determination of dissolved organic carbon. *Can. J. Fish. Aquat. Sci.*, 44(1),  
523 214-218.

524 Meybeck, M. (1982). Carbon, nitrogen, and phosphorus transport by world rivers. *American*  
525 *Journal of Science*, 282(4), 401–450. <https://doi.org/10.2475/ajs.282.4.401>

526 Micinski, E., Ball, L. A., & Zafiriou, O. C. (1993). Photochemical oxygen activation:  
527 Superoxide radical detection and production rates in the eastern Caribbean. *Journal of*  
528 *Geophysical Research: Oceans*, 98(C2), 2299–2306.  
529 [https://doi.org/10.1029/92JC02766@10.1002/\(ISSN\)2169-9291.PECW1](https://doi.org/10.1029/92JC02766@10.1002/(ISSN)2169-9291.PECW1)

530 Minella, M., De Laurentiis, E., Maurino, V., Minero, C., & Vione, D. (2015). Dark  
531 production of hydroxyl radicals by aeration of anoxic lake water. *Science of the Total*  
532 *Environment*, 527–528, 322–327. <https://doi.org/10.1016/j.scitotenv.2015.04.123>

533 Neubauer, E., Köhler, S. J., von der Kammer, F., Laudon, H., Hofmann, T., Neubauer, E., et  
534 al. (2013). Effect of pH and stream order on iron and arsenic speciation in boreal  
535 catchments. *Environmental Science & Technology*, 47(13), 1–14.  
536 <https://doi.org/10.1021/es401193j>

537 Page, S. E., Sander, M., Arnold, W. A., & McNeill, K. (2012). Hydroxyl radical formation  
538 upon oxidation of reduced humic acids by oxygen in the dark. *Environmental Science &*  
539 *Technology*, 46(3), 1590–1597. <https://doi.org/10.1021/es203836f>

540 Page, S. E., Kling, G. W., Sander, M., Harrold, K. H., Logan, J. R., McNeill, K., & Cory, R.  
541 M. (2013). Dark formation of hydroxyl radical in arctic soil and surface waters.  
542 *Environmental Science & Technology*, 47(22), 12860–12867.  
543 <https://doi.org/10.1021/es4033265>

544 Petasne, R. G., & Zika, R. G. (1987). Fate of superoxide in coastal sea water. *Nature*,  
545 325(6104), 516–518. <https://doi.org/10.1038/325516a0>

546 Pignatello, J. J., Oliveros, E., & MacKay, A. (2006). Advanced oxidation processes for  
547 organic contaminant destruction based on the fenton reaction and related chemistry.  
548 *Critical Reviews in Environmental Science and Technology*, 36(1), 1–84.  
549 <https://doi.org/10.1080/10643380500326564>



550 Raymond, P. A., Hartmann, J., Lauerwald, R., Sobek, S., McDonald, C., Hoover, M., et al.  
551 (2013). Global carbon dioxide emissions from inland waters. *Nature*, 503(7476), 355–  
552 359. <https://doi.org/10.1038/nature12760>

553 Rush, J. D., & Bielski, B. H. J. (2005). Pulse radiolytic studies of the reaction of  
554 perhydroxyl/superoxide O<sub>2</sub><sup>-</sup> with iron(II)/iron(III) ions. The reactivity of HO<sub>2</sub>/O<sub>2</sub><sup>-</sup> with  
555 ferric ions and its implication on the occurrence of the Haber-Weiss reaction. *The*  
556 *Journal of Physical Chemistry*, 89(23), 5062–5066. <https://doi.org/10.1021/j100269a035>

557 Salonen, K. (1979). A versatile method for the rapid and accurate determination of carbon by  
558 high temperature combustion1. *Limnology and Oceanography*, 24(1), 177–183.  
559 <https://doi.org/10.4319/lo.1979.24.1.0177>

560 Salonen, K. (1981). Rapid and precise determination of total inorganic carbon and some  
561 gases by high temperature combustion. *Water Research*, 15(15), 403–406.

562 Salonen, K., & Kononen, K. (1984). Applicability of size fractionation to assess respiration in  
563 different size classes of plankton. *Arch. Hydrobiol. Beih. Ergebn. Limnol.*, 19, 223–227.

564 Sekar, R., & DiChristina, T. J. (2014). Microbially driven fenton reaction for degradation of  
565 the widespread environmental contaminant 1,4-dioxane. *Environmental Science &*  
566 *Technology*, 48(21), 12858–12867. <https://doi.org/10.1021/es503454a>

567 Studenroth, S., Huber, S. G., Kotte, K., & Schöler, H. F. (2013). Natural abiotic formation of  
568 oxalic acid in soils: Results from aromatic model compounds and soil samples.  
569 *Environmental Science & Technology*, 47(3), 1323–1329.  
570 <https://doi.org/10.1021/es304208a>

571 Trusiak, A., Treibergs, L. A., Kling, G. W., & Cory, R. M. (2018). The role of iron and  
572 reactive oxygen species in the production of CO<sub>2</sub> in arctic soil waters. *Geochimica et*  
573 *Cosmochimica Acta*, 224, 80–95. <https://doi.org/10.1016/j.gca.2017.12.022>

574 Vähätalo, A. V., Aarnos, H., & Mäntyniemi, S. (2010). Biodegradability continuum and  
575 biodegradation kinetics of natural organic matter described by the beta distribution.  
576 *Biogeochemistry*, *100*(1–3), 227–240. <https://doi.org/10.1007/s10533-010-9419-4>

577 Vermilyea, A. W., Dixon, T. C., & Voelker, B. M. (2010). Use of H<sub>2</sub><sup>18</sup>O<sub>2</sub> to measure absolute  
578 rates of dark H<sub>2</sub>O<sub>2</sub> production in freshwater systems. *Environmental Science &*  
579 *Technology*, *44*(8), 3066–3072. <https://doi.org/10.1021/es100209h>

580 Vione, D., Minella, M., Maurino, V., & Minero, C. (2014). Indirect photochemistry in sunlit  
581 surface waters: Photoinduced production of reactive transient species. *Chemistry - A*  
582 *European Journal*, *20*(34), 10590–10606. <https://doi.org/10.1002/chem.201400413>

583 Voelker, B. M., Morel, F. M. M., & Sulzberger, B. (1997). Iron redox cycling in surface  
584 waters: Effects of humic substances and light. *Environmental Science & Technology*,  
585 *31*(4), 1004–1011. <https://doi.org/10.1021/es9604018>

586 Waggoner, D. C., Wozniak, A. S., Cory, R. M., & Hatcher, P. G. (2017). The role of reactive  
587 oxygen species in the degradation of lignin derived dissolved organic matter.  
588 *Geochimica et Cosmochimica Acta*, *208*, 171–184.  
589 <https://doi.org/10.1016/j.gca.2017.03.036>

590 Wolf, R., Thrane, J.-E., Hessen, D. O., & Andersen, T. (2018). Modelling ROS formation in  
591 boreal lakes from interactions between dissolved organic matter and absorbed solar  
592 photon flux. *Water Research*, *132*, 331–339.  
593 <https://doi.org/10.1016/J.WATRES.2018.01.025>

594 Xiao, Y., Hoikkala, L., Kasurinen, V., Tirola, M., Kortelainen, P., & Vähätalo, A. V. (2016).  
595 The effect of iron on the biodegradation of natural dissolved organic matter. *Journal of*  
596 *Geophysical Research: Biogeosciences*, 1–18. <https://doi.org/10.1002/2016JG003394>

597 Xiao, Y., Carena, L., Näsi, M.-T., & Vähätalo, A. V. (2020). Superoxide-driven autocatalytic  
598 dark production of hydroxyl radicals in the presence of complexes of natural dissolved

599 organic matter and iron. *Water Research*, 177, 115782. [https://doi.org/](https://doi.org/10.1016/j.watres.2020.115782)  
600 10.1016/j.watres.2020.115782.

601 Yuan, X., Davis, J. A., & Nico, P. S. (2016). Iron-mediated oxidation of  
602 Methoxyhydroquinone under dark conditions: Kinetic and mechanistic insights.  
603 *Environmental Science & Technology*, 50(4), 1731–1740.  
604 <https://doi.org/10.1021/acs.est.5b03939>

605 Zazo, J. A., Casas, J. A., Mohedano, A. F., Gilarranz, M. A., & Rodríguez, J. J. (2005).  
606 Chemical pathway and kinetics of phenol oxidation by Fenton’s reagent. *Environmental*  
607 *Science & Technology*, 39(23), 9295–9302. <https://doi.org/10.1021/es050452h>

608 Zepp, R. G., Faust, B. C., & Jürg, H. (1992). Hydroxyl radical formation in aqueous reactions  
609 (pH 3-8) of Iron(II) with hydrogen peroxide: The photo-Fenton reaction. *Environmental*  
610 *Science & Technology*, 26(2), 313–319. <https://doi.org/10.1021/es00026a011>

611 Zhang, T., Hansel, C. M., Voelker, B. M., & Lamborg, C. H. (2016). Extensive dark  
612 biological production of reactive oxygen species in brackish and freshwater ponds.  
613 *Environmental Science & Technology*, 50(6), 2983–2993.  
614 <https://doi.org/10.1021/acs.est.5b03906>  
615

616 **Tables**

617 Table 1. Experimental schemes. VK, Lake Valkea-Kotinen; IV, Lake Iso Valkjärvi.

Name of experiment	Sampling date original pH	Adjusted pH filtration container	Incubation conditions	Measured response
1- $\mu\text{m}$	26 October pH 5.4 VK, 5.3 IV	pH 4, 5, and 6 1- $\mu\text{m}$ ampoule	23°C 584 d dark	Mineralization
1- $\mu\text{m}$ or 10- $\mu\text{m}$	2 September pH 5.3 VK, 5.2 IV	pH 4 and 5 1- $\mu\text{m}$ or 10- $\mu\text{m}$ ampoule	23°C 244 d dark	Mineralization
0.1- $\mu\text{m}$ photochemistry	IV: 15 September VK: 17 September pH 5.5 VK, 5.2 IV	pH 4 and 5 0.1- $\mu\text{m}$ (autoclaved) ampoule	15°C 622 d or 623 d light or dark	Mineralization
1- $\mu\text{m}$ light or dark	IV: 15 September VK: 17 September pH 5.5 VK, 5.2 IV	pH 4 and 5 1- $\mu\text{m}$ ampoule	15°C 622 d or 623 d light or dark	Mineralization
Fe	26 October pH 5.4 VK	pH 4 and 5 SPE-DOM + microbial isolate	23°C 28 d dark	Bacterial density

618

619

620 Table 2. Parameters of the reactivity continuum model (Eq. 1 and 2). The values of  $a$  and  $v$   
 621 were estimated by fitting the Eq. 1 to the measured concentrations of OC shown as markers  
 622 in Figure 1.

Experiment	Treatment	Valkea-Kotinen			Iso Valkjärvi		
		$a$ (day)	$v$	$k_0, v/a$ (day <sup>-1</sup> )	$a$ (day)	$v$	$k_0, v/a$ (day <sup>-1</sup> )
1- $\mu\text{m}$ Dark, 23°C, 584 d	pH 4	64.5	0.138	0.0021	29.8	0.089	0.0030
	pH 5	38.1	0.083	0.0022	22.4	0.048	0.0021
	pH 6	53.1	0.088	0.0017	21.5	0.039	0.0018
1- $\mu\text{m}$ or 10- $\mu\text{m}$ Dark, 23°C, 244 d	pH 4 1- $\mu\text{m}$	94.6	0.109	0.0012	26.9	0.079	0.0029
	pH 5 1- $\mu\text{m}$	60.2	0.053	0.0009	18.2	0.034	0.0019
	pH 4 10- $\mu\text{m}$	68.4	0.594	0.0087	-	-	-
	pH 5 10- $\mu\text{m}$	28.0	0.139	0.0050	20.5	0.157	0.0077
0.1- $\mu\text{m}$ autoclaved Light, 15°C, 622 d or 623 d	pH 4 irradiated	145	0.095	0.0007	308	0.119	0.0004
	pH 5 irradiated	362	0.100	0.0003	204	0.067	0.0003
1- $\mu\text{m}$ Light or Dark, 15°C, 622 d or 623 d	pH 4 irradiated	252	0.382	0.0015	101	0.218	0.0022
	pH 5 irradiated	103	0.136	0.0013	71.2	0.095	0.0013
	pH 4 dark	199	0.113	0.0006	66.6	0.090	0.0014
	pH 5 dark	81.0	0.042	0.0005	128	0.081	0.0006

623 “-” not determined.

624

625 Table 3. Mineralization of OC (% of initial OC) induced by irradiation during 623 days in the  
 626 0.1- $\mu\text{m}$  autoclaved and in the 1- $\mu\text{m}$  filtered lake water (Table 2).

Lake and pH	Mineralization of OC induced by irradiation (%) <sup>†</sup>	
	0.1- $\mu\text{m}$ autoclaved	1- $\mu\text{m}$ with bacteria
Valkea-Kotinen pH 4	14.8 $\pm$ 0.5	22.6 $\pm$ 3.6
Valkea-Kotinen pH 5	9.6 $\pm$ 0.1	14.5 $\pm$ 0.6
Iso Valkjärvi pH 4	12.4 $\pm$ 0.3	16.0 $\pm$ 4.2
Iso Valkjärvi pH 5	9.1 $\pm$ 1.3	6.1 $\pm$ 2.8

627 <sup>†</sup>calculated as the difference between irradiated waters and their dark controls. Error  
 628 represents the standard deviations of replicated ( $n = 3$ ) irradiated and dark treatments.

629

630 Table 4. The contribution of acidification (from pH 5 to 4) to the mineralization of OC during  
 631 20 months through microbes (1- $\mu\text{m}$  or 10- $\mu\text{m}$  filtrates in the dark), photochemistry and the  
 632 combined action of photochemistry and microbes in two lake waters.

Category	Experiment	% mineralized by acidification*			
		Valkea-Kotinen		Iso Valkjärvi	
		$\mu\text{mol L}^{-1}\dagger$	Fraction (%) $\ddagger$	$\mu\text{mol L}^{-1}\dagger$	Fraction (%) $\ddagger$
Biological in the dark					
1- $\mu\text{m}$	1- $\mu\text{m}$	53	24	58	38
	1- $\mu\text{m}$ or 10- $\mu\text{m}$	61	39	63	49
10- $\mu\text{m}$	1- $\mu\text{m}$ or 10- $\mu\text{m}$	313	53		
Photochemical (abiotic irradiated)					
	0.1 $\mu\text{m}$ autoclaved	41	35	20	27
Biological+Photochemical (biological irradiated)					
	1 $\mu\text{m}$ irradiated	113	38	89	44

633 \* Calculated according to the amount of mineralized OC during 20 months using the RC model (Eq. 1)  
 634 and values of  $a$  and  $v$  given in Table 2.

635  $\dagger$ OC mineralized by acidification ( $\mu\text{mol L}^{-1}$ ) = mineralized OC at pH 4 - mineralized OC at pH 5.

636  $\ddagger$ % mineralized by acidification = 100 (mineralized OC at pH 4 – mineralized OC at pH  
 637 5)/mineralized OC at pH 4.

638