



HAL
open science

Fenton oxidation to remediate PAHs in contaminated soils: A critical review of major limitations and counter-strategies

Muhammad Usman, Khalil Hanna, Stefan B. Haderlein

► **To cite this version:**

Muhammad Usman, Khalil Hanna, Stefan B. Haderlein. Fenton oxidation to remediate PAHs in contaminated soils: A critical review of major limitations and counter-strategies. *Science of the Total Environment*, 2016, 569–570, pp.179–190. 10.1016/j.scitotenv.2016.06.135 . hal-01338529

HAL Id: hal-01338529

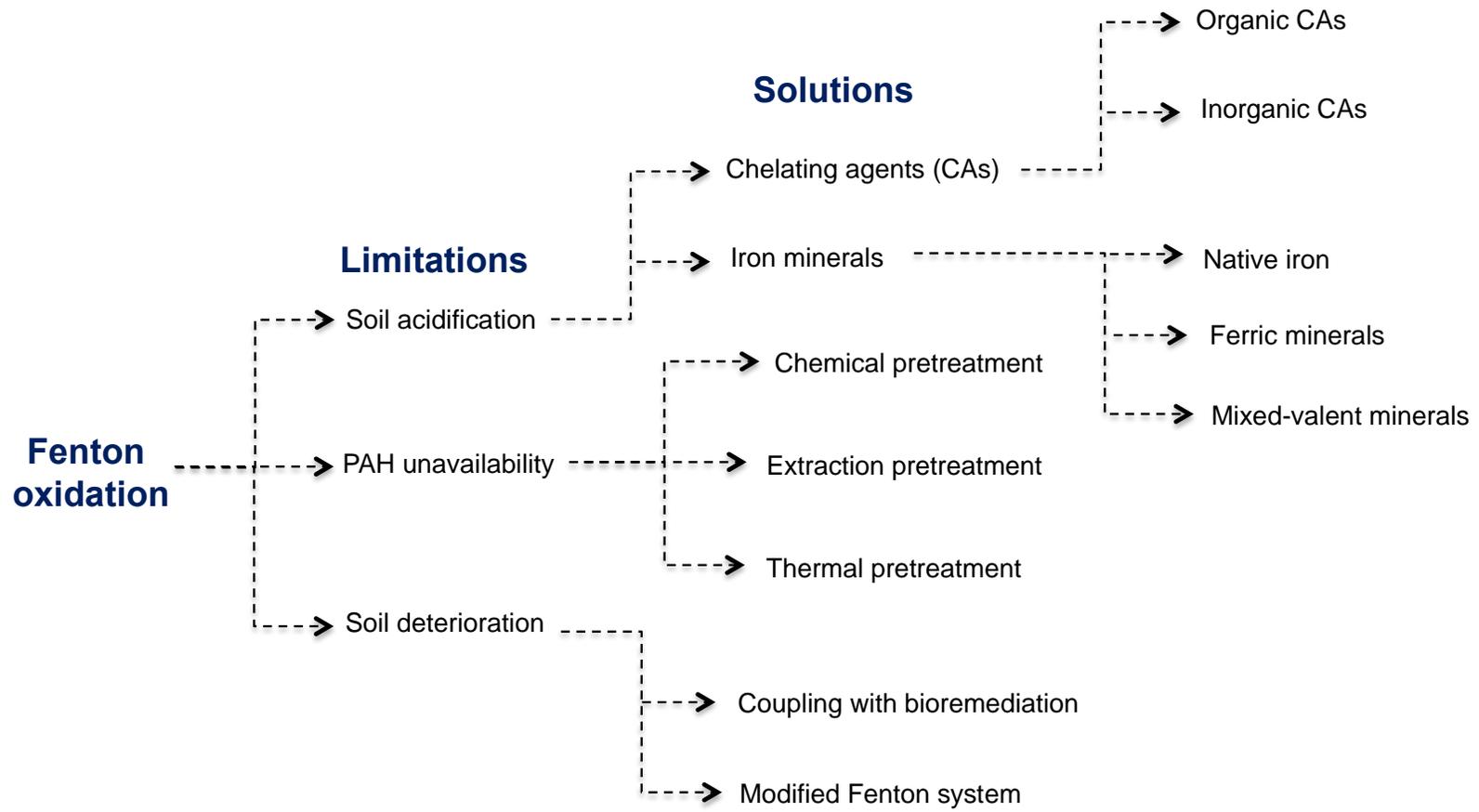
<https://univ-rennes.hal.science/hal-01338529>

Submitted on 5 Dec 2016

HAL is a multi-disciplinary open access archive for the deposit and dissemination of scientific research documents, whether they are published or not. The documents may come from teaching and research institutions in France or abroad, or from public or private research centers.

L'archive ouverte pluridisciplinaire **HAL**, est destinée au dépôt et à la diffusion de documents scientifiques de niveau recherche, publiés ou non, émanant des établissements d'enseignement et de recherche français ou étrangers, des laboratoires publics ou privés.

Graphical Abstract



Highlights

- 1st comprehensive review of major limitations of PAH removal by Fenton oxidation.
- Limitations under review are acidic pH, PAH unavailability and soil deterioration.
- Critical review of various strategies to address these limitations is presented.
- Gaps and future research needs are highlighted for application at field scale.

1 **Fenton oxidation to remediate PAHs in contaminated soils: A critical review of major**
2 **limitations and counter-strategies**

3
4 M. Usman^{a, b*}, K. Hanna^c, S. Haderlein^a

5
6 ^a Department of Geosciences, Center for Applied Geosciences, University of Tübingen,
7 72074 Tübingen, Germany

8 ^b Institute of Soil and Environmental Sciences, University of Agriculture, Faisalabad 38040,
9 Pakistan

10 ^c Ecole Nationale Supérieure de Chimie de Rennes, UMR CNRS 6226, 11 Allée de Beaulieu,
11 35708 Rennes Cedex 7, France

12
13
14
15
16 * Corresponding author:

17 Dr. M. Usman

18 Research Fellow (Alexander von Humboldt Foundation)

19 Department of Geosciences, Center for Applied Geosciences, University of Tübingen

20 Hölderlinstr. 12, 72074 Tübingen, Germany

21 Email: muhammad.usman@uaf.edu.pk or: musmanch@yahoo.com

22 Phone No.: +49 70 71 29 78 924

23 Fax: +49 70 71 29 50 59

24 **Abstract**

25 Fenton oxidation constitutes a viable remediation strategy to remove polycyclic aromatic
26 hydrocarbons (PAHs) in contaminated soils. This review is intended to illustrate major
27 limitations associated with this process like acidification, PAH unavailability, and
28 deterioration of soil quality along with associated factors, followed by a critical description of
29 various developments to overcome these constraints. Considering the limitation that its
30 optimal pH is around 3, traditional Fenton treatment could be costly, impractical in soil due
31 to the high buffering capacity of soils and associated hazardous effects. Use of various
32 chelating agents (organic or inorganic) allowed oxidation at circumneutral pH but factors like
33 higher oxidant demand, cost and toxicity should be considered. Another alternative is the use
34 of iron minerals that can catalyze Fenton-like oxidation over a wide range of pH, but mobility
35 of these particles in soils (i.e. saturated and unsaturated zones) should be investigated prior to
36 *in-situ* applications. The PAH-unavailability is the crucial limitation hindering their effective
37 degradation. Research data is compiled describing various strategies to address this issue like
38 the use of availability enhancement agents, extraction or thermal pretreatment. Last section of
39 this review is devoted to describe the effects of various developments in Fenton treatment
40 onto soil quality and native microbiota. Finally, research gaps are discussed to suggest future
41 directions in context of applying Fenton oxidation to remediate contaminated soils.

42

43 **Keywords**

44 PAH Remediation; Fenton oxidation; pH; PAH unavailability; Soil quality

45 **1. Introduction**

46 Polycyclic aromatic hydrocarbons (PAHs) are organic compounds composed of two or more
47 fused benzene rings. Their contamination in environment is of great concern due their high
48 toxicity, carcinogenic effects and environmental persistence. There are sixteen PAHs that are
49 considered as priority pollutants by United States Environmental Protection Agency (US
50 EPA) and European community. For being recalcitrant in nature, PAHs tend to persist in the
51 environment resulting in a ubiquitous distribution. The European Environment Agency
52 (EEA) located about 1.5 millions of contaminated sites in Europe including 200,000 sites
53 contaminated PAHs (E.E.A, 2012). Soil often acts as the ultimate depository of PAHs
54 particularly due to their hydrophobic nature (Haritash and Kaushik, 2009). Owing to the
55 strong persistence and ubiquitous occurrence, substantial research has been devoted to
56 develop techniques to remove persistent PAHs from complex matrix like soil or sediments.
57 Chemical oxidation has emerged as a viable remediation strategy to degrade PAHs in
58 contaminated soils (Lee et al., 1998; Watts et al., 2002; Rivas, 2006; Ferrarese et al., 2008;
59 Yap et al., 2011). The most common oxidants used in this technique are permanganate,
60 ozone, persulfate and Fenton's reagent (H₂O₂ combining with iron) (ITRC, 2005). Among
61 these oxidants, Fenton oxidation is showing promising potential to rapidly degrade PAHs in
62 contaminated soils. Fenton treatments utilize the high reactivity of hydroxyl radical (·OH), a
63 very reactive chemical species, which is generated by the decomposition of hydrogen
64 peroxide (H₂O₂) by iron through following reaction (Fenton, 1894):



66 Over the last two decades, use of Fenton oxidation has been well documented to treat PAHs
67 in contaminated soils by numerous research studies. Apart from research data, few reviews
68 focusing on chemical oxidation/Fenton-based treatments specifically for PAHs alone or along
69 with other persistent organic pollutants in contaminated soils have been published (Rivas,

70 2006; Gan et al., 2009; Yap et al., 2011; Venny et al., 2012a). First review about chemical
71 oxidation of PAHs was completed by F.J. Rivas (Rivas, 2006) where he focused mainly on
72 the Fenton oxidation, ozone and high temperature pressurized water after a detailed
73 discussion about sorption and extraction of PAHs from contaminated soils. Various
74 techniques available for the remediation of PAH contaminated soils were reviewed by Gan
75 and co-workers (2009) including bioremediation, solvent extraction, bioremediation,
76 phytoremediation, chemical oxidation, photocatalytic degradation, electrokinetic remediation,
77 thermal treatment and their integration. Yap and co-workers (2011) presented the first review
78 specifically for the application of Fenton-based treatments (conventional, modified, Fenton-
79 like along with integrated approaches) for PAHs-contaminated soils. Review by Venny and
80 co-workers (2012a) described the fundamental principles and governing factors of Fenton
81 oxidation to remediate soils contaminated with PAHs, polychlorinated biphenyls (PCBs),
82 polychlorinated dibenzo-p-dioxins and furans and coupling of Fenton oxidation with other
83 strategies like bioremediation and eletrokinetic remediation.

84 To the best of our knowledge, a systematic review illustrating the major limitations of Fenton
85 treatment along with proposed solutions to overcome these constraints is still missing in
86 literature. Thus, this review article is intended to collect research data related to potential
87 constraints of Fenton oxidation in context of PAH contaminated soils. In addition, recent
88 developments to overcome these limitations are critically evaluated along with associated
89 factors in this review. These drawbacks would depend on the remedial strategy and scale, and
90 the targeted zone in soils (e.g. aquifer or vadose zone). Despite the large amount of published
91 studies related to Fenton treatments for PAH-contaminated soils at lab-scale (Yap et al.,
92 2011), very limited data is available on the remediation performance at pilot/field scale (Liu
93 et al., 1993; Brown, 1997; Lingle et al., 2001; ITRC, 2005; Taylor, 2012). Prior to the field or
94 *in-situ* applications, feasibility studies are generally conducted at batch scale to determine the

95 extent and rate of PAH removal. However, few reports have investigated the Fenton
96 oxidation under column dynamic conditions, which more closely mimic field conditions
97 (Palmroth et al., 2006a; Laurent et al., 2012; Venny et al., 2012c; Lemaire et al., 2013b;
98 Venny et al., 2014). Figure 1 shows the different experimental setups used to investigate
99 Fenton oxidation in contaminated soils at laboratory scale. Column studies, which are better
100 representative of real field conditions, can be carried out under water-saturated (that
101 represents aquifer) and water-unsaturated conditions (that represent vadose zone).
102 We are convinced that this review article will be of significant interest for researchers
103 working on chemical oxidation of PAHs-contaminated soils. Finally, the literature lack of
104 information and data that is crucial prior to *in-situ* field applications is also highlighted.

105

106 **2. Major limitations of Fenton oxidation in PAH-contaminated soil**

107 Fenton oxidation has validated its potential as viable remediation technique for PAH-
108 contaminated soils but there are various limitations associated with this process. This review
109 article focuses on the three major limitations of this technique in context of PAH-
110 contaminated soil including: i) acidification of medium, ii) PAH unavailability and iii)
111 deterioration of soil quality, fertility and microbial community. These limitations are
112 discussed in the following sections along with a critical discussion of their solutions proposed
113 in literature.

114

115 **2.1. Acidification of reaction medium**

116 Traditional Fenton's reagent ($\text{Fe}^{\text{II}}\text{-H}_2\text{O}_2$) displays its maximum oxidation activity at a pH
117 about 2.8 – 3 (Safarzadeh-Amiri et al., 1996; Pignatello et al., 2006). Thus, most of the
118 remediation studies to remove PAHs in contaminated soils employed traditional Fenton
119 reagent at acidic pH (Yap et al., 2011) which is probably the most important limitation of this

120 process. This initial acidification (optimum pH is 3) is required to keep Fe^{II} in its soluble
121 form (Tamura et al., 1980). The iron precipitation occurring after pH 4.0 led to the
122 obstruction of activation processes (Usman et al., 2012), through several parasite reactions
123 including the decomposition of H_2O_2 to oxygen. In addition, sorption of Fe^{II} or Fe^{III} ions onto
124 mineral or organic constituents in soil increases with increase in pH (Hanna et al., 2013;
125 Catrouillet et al., 2014) and thus, rendering the Fe species less available for Fenton
126 activation.

127 This acidification step is operated not only to keep Fe in dissolved form but also because the
128 $\bullet\text{OH}$ generation in reaction (1) is highest at pH 3 that usually decreases at higher or lower pH
129 values (Gallard et al., 1998). Furthermore, reaction (1) actually involves the formation of a
130 highly oxidised Fe adduct, often indicated as ferryl (e.g. ferryl ion FeO^{2+}), which does not
131 necessarily evolve into $\bullet\text{OH}$ (Pignatello et al., 1999).

132 Initial acidification required for optimum efficiency of Fenton's reagent is costly and
133 impractical in soil system due to buffering capacity of soils (Ershadi et al., 2011). This could
134 also create a hazardous situation for soil quality and microbes in soil buffered systems, where
135 a circumneutral pH prevails (Nam et al., 2001; Sahl and Munakata-Marr, 2006; Sirguyey et al.,
136 2008; Valderrama et al., 2009). Therefore, traditional Fenton oxidation was found
137 incompatible with subsequent biological treatment to remediate PAH contaminated soil (Nam
138 et al., 2001). It may also increase mobilization of heavy metals that would devastate the soil
139 ecosystem. Acidic pH along with high amount of Fe^{II} was also linked to a remarkable
140 increase in available Fe and Al in treated soil, which reached the toxicity level (Gan et al.,
141 2013). This decrease in pH during Fenton oxidation was also found as the major limiting
142 factor for subsequent vegetation in treated soil (Sirguyey et al., 2008). It was reported by
143 Valderrama and co-workers (2009) that use of more aggressive traditional Fenton oxidation
144 does not favor the subsequent biological treatment which was explained by the residual

145 oxidant that may remain in the soil and inhibit the biodegradation after the application of
146 higher oxidant dose.

147 Therefore, various strategies were considered in literature that allows chemical oxidation at
148 circumneutral pH but without compromising the oxidation efficiency for efficient PAH
149 degradation like the use of chelating agents or iron minerals. These strategies will be detailed
150 in the following paragraphs.

151

152 **2.1.1 Use of Chelating agents**

153 Use of chelating agents along with traditional Fenton treatment allows chemical oxidation at
154 circumneutral pH and the reaction is termed as modified Fenton (MF) oxidation (Nam et al.,
155 2001). Chelating agents (CAs) are mostly organic compounds that bind iron and therefore
156 maintain in its dissolved form. By binding with Fe^{II} or Fe^{III} , CAs can maintain their ability to
157 catalyze H_2O_2 at circumneutral pH. These chelating agents are used to extend the
158 applicability of Fenton oxidation to native soil pH that has more practical implications for *in-*
159 *situ* remediation. Different kinds of chelating agents have been explored in MF (Table 1)
160 including cyclodextrines (CD), ethylenediaminetetraacetic acid (EDTA), catechol (CC),
161 gallic acid (GA), citric acid (CA), oxalic acid (OA), humic acid (HA), malic acid (MA),
162 sodium citrate (SC) and other amino-poly-acetic acids (Nam et al., 2001; Gryzenia et al.,
163 2009; Veignie et al., 2009; Venny et al., 2012b).

164 For example, Nam et al. (Nam et al., 2001) showed that the addition of organic CAs (such as
165 CC and GA) along with ferric ion during MF oxidation (pH 6.0–6.5) could have distinct
166 advantages due to its compatibility with subsequent bioremediation as compared to the
167 traditional Fenton oxidation at pH 2–3. Their results revealed that >80% of low molecular
168 weight (LMW) PAHs and only 20-40% of high molecular weight (HMW) PAHs were
169 degraded during traditional Fenton oxidation. In their attempt to overcome the limitations of

170 conventional Fenton oxidation, combined MF and biodegradation resulted in > 98% removal
171 of LMW-PAHs and > 70% removal of HMW-PAHs. In their attempt to examine HA as iron
172 chelator in MF oxidation, Georgi and co-workers (2007) reported that presence of HA (50–
173 100 mg L⁻¹) in MF system greatly enhanced the degradation of organic compounds in the
174 range of pH 5–7. They also highlighted that similar degradation extent can be obtained by
175 MF oxidation at pH 5 in the presence of HA and traditional Fenton process at pH 3 in the
176 absence of HA (Georgi et al., 2007).

177 In addition to binding Fe^{II} ion for enhanced solubility at circumneutral pH, CD (Veignie et
178 al., 2009) or HA (Georgi et al., 2007) also resulted in improving availability of hydrophobic
179 organic pollutants in contaminated soils by forming complexes with pollutants. Formation of
180 pollutant-CD-iron or pollutant-HA-iron complex in microenvironment allows hydroxyl
181 radicals to specifically target on pollutants by eliminating the detrimental soil matrix effect
182 for improved oxidation efficiency (Lindsey et al., 2003; Georgi et al., 2007; Veignie et al.,
183 2009). Owing to the ability of CD to improve pollutant availability, it was also employed in
184 soil washing solution (Viglianti et al., 2006), bioremediation (Bardi et al., 2000) and
185 chemical oxidation (Usman et al., 2012) that improved the removal efficiency many times. A
186 detailed discussion about the use of CD to improve PAH availability is presented below in
187 Section 2.2.1 (Chemical Pretreatment).

188 Despite the promising efficiency of chelating agents to promote pollutant oxidation, their use
189 is limited by cost, toxicity and potential negative effects on oxidation efficiency due to the
190 non-productive consumption of the CAs (Xue et al., 2009b; Pardo et al., 2014). For being
191 organic in nature, CAs may compete with target pollutants for oxidant and thus act as radical
192 scavenger by increasing oxidant demand (Georgi et al., 2007; Xue et al., 2009b; Pardo et al.,
193 2014). Oxidation of CAs themselves will thereby lead to the loss of their chelating ability
194 which is required to enhance pollutant degradation (Georgi et al., 2007). Moreover, chelating

195 agents like EDTA can also pose environmental risks because of their persistence in the
196 environment and their contribution to improve heavy metal mobility/bioavailability
197 (Sillanpaa and Pirkanniemi, 2001; Yang et al., 2001; Rastogi et al., 2009).

198 Recently inorganic CAs like sodium pyrophosphate (SP) were proposed as an alternative in
199 MF oxidation of PAHs in contaminated soils at circumneutral pH (Venny et al., 2012b;
200 Venny et al., 2012c; Jorfi et al., 2013). Comparison of various organic and inorganic CAs
201 (Fig. 2a) to assist MF oxidation revealed the following efficiency order for PAH degradation:
202 $SP > EDTA > OA > MA > SC$ (Venny et al., 2012b). Other studies by this research group
203 have established the higher efficiency of SP during Fenton oxidation and its compatibility
204 with biological treatment to promote PAH degradation (Venny et al., 2012c; Venny et al.,
205 2014). Similarly, Jorfi et al. (Jorfi et al., 2013) performed Fenton oxidation by employing
206 iron nano-particles along with various CAs to remove pyrene in contaminated soil. They
207 reported that removal efficiency was found to be maximum for SP (93%) compared to 86%,
208 75%, 72% and 71% for EDTA, SC, HA and FA, respectively at H_2O_2/Fe^{III} molar ratio of 10,
209 pH 7 and the contact time of 6 h. Use of inorganic CAs could be advantageous since they are
210 less likely to act as radical scavenger and do not increase the total organic carbon of system
211 (Rastogi et al., 2009) although prolonged treatment could make SP to act as radical scavenger
212 (Venny et al., 2012c). In addition to less oxidant scavenging, decomposition of H_2O_2 with SP
213 was lower than with organic CAs (Fig. 2b) underscoring higher stability of oxidant when
214 mixed with inorganic SP (Venny et al., 2012b).

215

216 **2.1.2. Use of Fe-bearing minerals**

217 In order to overcome the drawback of acidification employed in traditional Fenton oxidation,
218 iron mineral can be used instead of soluble Fe^{II} at circumneutral pH, and the reaction is
219 defined as a Fenton-like (FL) reaction. Iron minerals are able to catalyze Fenton-like

220 reactions over a wide range of pH values (Watts et al., 2002; Hanna et al., 2008; Jung et al.,
221 2009). The water solubility of Fe-oxyhydroxides as well as the possible leaching of iron from
222 their surface is low. The Fe-minerals catalyze oxidation mainly through heterogeneous
223 reaction, i.e. activation of oxidant by Fe surface sites located on oxide surfaces. As a result,
224 the ability of Fe to catalytically decompose H_2O_2 and generate hydroxyl radicals can be
225 maintained with respect to traditional Fenton, but the overall oxidation kinetics is generally
226 lower (Garrido-Ramírez et al., 2010). Various mechanisms are proposed for oxidative
227 degradation of organic compounds on the surface of iron oxide catalysts during FL oxidation
228 and are detailed elsewhere (Xue et al., 2009a; Garrido-Ramírez et al., 2010; Yap et al., 2011).
229 There are various kinds of iron minerals including ferric- Fe^{III} (ferrihydrite, goethite,
230 lepidocrocite, hematite etc.) and mixed Fe^{II} - Fe^{III} oxides (magnetite and green rust) (Cornell
231 and Schwertmann, 2006). Heterogeneous catalysts including native or added iron oxides or
232 certain transition metals can catalyze the FL oxidation in soils at circumneutral pH (Flotron et
233 al., 2005; Usman et al., 2012), which may be especially advantageous for *in situ* remediation
234 of contaminated soils where pH cannot be adjusted.

235 It was also reported by Jung and co-workers (2009) that H_2O_2 was more stable (as it has a ten
236 times longer life time) under FL reaction (catalyzed by magnetite or hematite) than that with
237 soluble Fe^{II} . Moreover, mineral-catalyzed oxidation system (3 ml of 30% H_2O_2 for 5 g
238 contaminated sand) without pH adjustment showed better degradation (75.3%) of
239 phenanthrene sorbed on sand ($200\text{ mg}\cdot\text{kg}^{-1}$) than by Fenton activated by Fe^{II} (52.2%) (Jung et
240 al., 2009).

241 Remediation of PAHs in contaminated soils was reported by Fenton-like oxidation catalyzed
242 by native iron (Watts et al., 2002; Flotron et al., 2005) or iron minerals added into the system
243 like goethite (Kanel et al., 2003; Kanel et al., 2004), hematite (Jung et al., 2009), iron
244 oxyhydroxide/sodium dodecyl sulfate composites (Park and Kim, 2013) and magnetite

245 (Usman et al., 2012; Usman et al., 2013; Biache et al., 2015). Magnetite was found more
246 effective than goethite and soluble Fe^{II} at acidic pH to remediate petroleum hydrocarbons in
247 diesel and kerosene contaminated soil (Kong et al., 1998). Higher catalytic efficiency of
248 magnetite, a mixed Fe^{II}-Fe^{III} oxide, as compared to the ferric minerals was correlated to its
249 structural Fe^{II} (Matta et al., 2007; Hanna et al., 2008).

250 It was shown by Watts and co-workers (2002) that iron mineral present in solid matrix can
251 allow the production of reactive oxidant species for strong degradation of PAHs. They
252 reported that 85% of benzo[a]pyrene was removed when H₂O₂ (15 M) was catalyzed by
253 naturally occurring iron minerals (33.52 mg.g⁻¹ of soil) while a degradation of 70% was
254 achieved when H₂O₂ (15 M) was catalyzed by soluble Fe^{II} (6.6 mM) in sand. Role of
255 endogenous iron minerals was highlighted when application of sole H₂O₂ resulted in PAH
256 degradation almost equivalent to that obtained with H₂O₂ + soluble Fe^{II} without pH
257 adjustment in sludge (pH 8.8) and sediment (pH 4.4) bearing 94 mg.g⁻¹, 30 mg.g⁻¹ and not
258 determined total iron, respectively (Flotron et al., 2005). They reported similar findings in
259 contaminated soil (pH 7.1) where Fe contents were not determined. Owing to the presence of
260 native iron (16.4 mg.g⁻¹), application of Fenton oxidation (0.4 g H₂O₂.g⁻¹ soil) under
261 saturated conditions resulted in significant degradation of PAHs (upto 52%) without any
262 ferrous iron supplementation or initial acidification of soil (Palmroth et al., 2006a). Pardo and
263 co-workers (2016) observed similar degradation of PAHs by Fenton oxidation (882 mmol.L⁻¹
264 without pH adjustment) obtained by adding 1 mmol L⁻¹ Fe^{III} or without external addition of
265 Fe highlighting the role of endogenous Fe. They noted a similar concentration of iron in
266 solution after external addition of 1 mmol L⁻¹ Fe^{III} or iron extracted from soil (Pardo et al.
267 2016)

268 Kanel and co-workers (2004) reported that goethite can successfully catalyze FL oxidation
269 (33.5 g goethite.kg⁻¹ of soil and 5 M H₂O₂) with degradation extent of 73, 60, and 55% for

270 phenanthrene, anthracene, and pyrene, respectively in spiked sandy soil obeyed pseudo-first
271 order kinetics at natural pH. Use of magnetite (Fig. 3) to catalyze FL oxidation (H_2O_2 :Fe of
272 20, 10% w/w magnetite) at circumneutral pH (7 – 8) resulted in PAH degradation extent of
273 90% and 60% in spiked sandy soil and in historically contaminated soils (sandy loamy type)
274 sampled from former coking plant site, respectively (Usman et al., 2012). Under the similar
275 experimental conditions, soluble Fe^{II} was found unable to catalyze H_2O_2 as similar oxidation
276 efficiency (20% in spiked sandy soil and 5% in real soil) was observed when H_2O_2 was used
277 alone without iron activation and that was correlated to the precipitation of Fe^{II} at
278 circumneutral pH in tested soils. Lower degradation in real soil samples as compared to the
279 spiked sandy soil was related to the soil matrix effect. Magnetite-catalyzed FL oxidation
280 (oxidant/Fe molar ratio equal to 10:1) also resulted in strong PAH removal (60–70 %) under
281 flow through conditions (flow rate 0.1 mL min^{-1} under water-saturated columns) in spiked
282 sandy soil having 10% of magnetite w/w (Usman et al., 2013). Biache and co-workers (2015)
283 also highlighted the strong efficiency of magnetite-catalyzed FL oxidation (14.5 mL of H_2O_2
284 (30%wt), 500 mg magnetite for 5 g soil) to degrade PAHs in three contaminated soils
285 sampled from former coking plant site (20% removal), former gas plant site soil (92%) and
286 active wood treatment facility (50%). Recently, pyrite (FeS_2) showed strong reactivity to
287 catalyze FL oxidation of PAHs in soil washing wastewater at circumneutral pH (Choi et al.,
288 2014). They reported a sustainable reactivity of this process that was linked to slow but
289 continuous Fe^{II} dissolution from the pyrite surface leads to the continuous formation of
290 reactive radicals during the oxidation reaction (Choi et al., 2014).

291 Pardo and co-workers (2016) recently conducted a study to optimize the application of the
292 Fenton oxidation to remediate PAH contaminated soils. They evaluated the suitability of
293 nZVI particles and Fe^{III} salt at different concentrations of initial iron ($1\text{--}5 \text{ mmol L}^{-1}$) and
294 oxidant ($88\text{--}1765 \text{ mmol L}^{-1}$) without pH adjustment. Almost similar degradation of PAHs

295 was achieved for the majority of the experiments. But, they reported that aqueous
296 concentration of iron was lower in case of nZVI and they linked to its lower solubility as
297 compared to Fe^{III} salt at circumneutral pH. Owing to this behavior, Pardo and co-workers
298 (2016) suggested that nZVI could act as a continuous Fe source that can be advantageous for
299 real field application.

300 Regardless of the investigated Fe-bearing minerals or nZVI particles to promote the *in-situ*
301 application of the oxidation treatment, profound investigations are required to evaluate
302 injection mode of these particles in soils. It will depend on the targeted area in soil whether
303 saturated aquifer (pump and treat) or unsaturated zone such as vadose zone (e.g. lysimeter
304 system). It is known that the efficiency of treatment significantly depends on the contact
305 “pollutant/catalyst/oxidant” (Lemaire et al., 2013a), and therefore particles with a high
306 mobility must easily reach the contaminated targeted zones (Johnson and Amy, 1995). This
307 mobility is strongly related to the composition of particles including functionalized or not,
308 particle size (micro- or nano-sized), concentration of suspension, etc. (Jiang et al., 2010).
309 Despite the mobility and colloidal transport of natural particles and some manufactured
310 nanoparticles have been already studied (Hannah and Thompson, 2008; Lowry et al., 2012),
311 there is limited information on the transport and fate of the catalyst particles used for
312 remediation purposes.

313 Apart from these strategies, few studies reported the use of Fenton’s reagent (H₂O₂ + soluble
314 Fe^{II}) to remove PAHs without adjusting the soil’s pH to avoid this initial acidification but that
315 leads to the partial degradation of PAHs (Kulik et al., 2006; Silva et al., 2009; Laurent et al.,
316 2012).

317

318

319

320 **2.2. PAH unavailability**

321 Another major constraint associated with Fenton oxidation in aged contaminated soils is the
322 poor availability of PAHs for remediation (Flotron et al., 2005; Usman et al., 2012; Lemaire
323 et al., 2013a; Choi et al., 2014) due to their strong hydrophobicity and partitioning into soil
324 constituents especially organic matter (Riding et al., 2013). This is caused by the deep
325 diffusion of the contaminants into the organic material that renders them unavailable for the
326 hydroxyl radicals, which are generated in the aqueous phase (Bogan et al., 2003). A strong
327 connection was reported between PAH availability and remediation efficiency of Fenton
328 oxidation (Usman et al., 2013; Biache et al., 2015; Usman et al., 2016). As a matter of fact,
329 magnetite-catalyzed FL oxidation was not effective unless PAH availability was enhanced in
330 two aged coking plant soils (Usman et al., 2012). Due to this strong correlation, Biache and
331 co-workers (2015) proposed that efficiency of Fenton based oxidation could serve as a valid
332 method to estimate PAH availability in contaminated soils. In order to suppress this
333 limitation and thus effectively degrade PAHs in contaminated soils, various strategies were
334 considered and are discussed below.

335

336 **2.2.1 Chemical pretreatment**

337 Chemical pretreatment involves the use of various chemical reagents to enhance the PAH
338 availability in contaminated soils as a pretreatment for or in combination with Fenton
339 oxidation. Four types of enhancement-solubility agents have been used including co-solvent,
340 surfactant, cyclodextrin and vegetable oil to solubilize PAHs. Co-solvents like ethanol were
341 used to increase solubility of PAH in soils that ultimately reduces the surface tension between
342 the compounds in soil and aqueous phase, and increase in PAH availability (Bonten et al.,
343 1999).

344 Ethanol-Fenton (ETF) treatment resulted in significant increase in PAH removal of 11–28%
345 (Lundstedt et al., 2006), 97% (Lee et al., 1998) and 63-93% (Lee et al., 2002) than by Fenton
346 alone in contaminated soils (Fig. 4). Application of ETF treatment in soil spiked with
347 benz(a)anthracene (500 mg/Kg of soil) using optimized experimental conditions (1.0 ml of
348 ethanol, 0.2 ml of 0.5 M Fe^{II}, and 0.3 ml of 30% H₂O₂ per g of soil) removed 97% of target
349 PAH and that neither water nor sodium dodecyl sulfate (SDS as surfactant) show a
350 substantial effect on removal (Lee et al., 1998). Ethanol was widely used as a solvent because
351 of its safety, low cost, and excellent extraction ability (Lee and Hosomi, 2001a).

352 Recently, Yap and co-workers (2012) reported the use of a novel system, ethyl lactate-Fenton
353 (ELF) treatment for soils artificially contaminated with four PAHs (500 mg/kg each of
354 phenanthrene, anthracene, fluoranthene and benzo[a]pyrene). Their results (Fig. 4) indicate
355 that ELF ([H₂O₂] = 666 mM, Fe^{II} = 66.6 mM with ethyl lactate, pH = 3) resulted in higher
356 PAH degradation (up to 97%) than ETF (70%) while conventional Fenton oxidation (without
357 ethanol or ethyl lactate) resulted in 31–57% of untreated PAHs. A recent study by the same
358 research group revealed the feasibility of using ELF to treat PAHs in aged soils (pH 4.4 and
359 6.2) having various quantities of soil organic carbon (Yap et al., 2015). Similar trend of
360 higher PAH degradation by ELF (H₂O₂ = 667.6 mM, Fe^{II} = 16.7 & 66.7 mM with varying
361 amounts of ethyl lactate) as compared to ETF or traditional Fenton (Fig. 4) was also indicated
362 elsewhere (Gan et al., 2013). However, addition of co-solvents could change the general non-
363 selective behavior of ·OH radicals towards selectivity as preferential degradation of
364 anthracene > benzo[a]pyrene > phenanthrene > fluoranthene was observed by ELF and ETF
365 treatments (Yap et al., 2012). They also suggested that degree of selectivity is co-solvent
366 dependent as selectivity was more pronounced in the ETF treatment than in the ELF
367 treatment. Moreover, presence of co-solvents during Fenton oxidation could also inhibit PAH
368 degradation by acting as radical scavenger (Qiang et al., 2000; Lundstedt et al., 2006; Yap et

369 al., 2012). But scavenging effect was more pronounced for EL than ET and it was correlated
370 to the higher bond ratio of C–O to C–H and higher oxidation state of EL (Yap et al., 2012).
371 This scavenging effect associated with ethanol was avoided by its use as a pretreatment step
372 (5 mL of ethanol/g soil for 24 h) where it was allowed to evaporate before soil samples were
373 subjected to Fenton-like oxidation (Usman et al., 2012). Treating ethanol pre-treated soil by
374 Fenton-like (FL) oxidation (magnetite + H₂O₂) caused 20% removal of PAHs (Fig. 5). It is
375 worthy to note that no PAH degradation was observed when FL oxidation was used directly
376 without any chemical pre-treatment (Usman et al., 2012). On the other hand, FL oxidation
377 after pretreatment using cyclodextrin (CD, 10 mM, overnight mixing) resulted in slightly
378 lower PAH degradation (15%) as compared to ethanol pretreatment (Usman et al., 2012)
379 probably due to scavenging effect of CD (Hanna et al., 2005). This scavenging effect
380 concerns only CD in study by Usman and co-workers (2012), as ethanol was allowed to
381 evaporate from the reactor before applying oxidants.

382 Use of surfactants was also found effective to improve the PAH availability decreasing the
383 interfacial tension and increasing their partitioning to the hydrophobic cores of surfactant
384 micelles. First use of surfactants in combination with Fenton's reagent was reported by
385 Martens and Frankenberger (Martens and Frankenberger Jr, 1995) who found that use of
386 sodium dodecyl sulfate (SDS) before Fenton oxidation resulted in higher PAH degradation
387 (30-45%) compared to without SDS. Application of cationic surfactant cetylpyridinium
388 chloride (CPC) in the pyrite Fenton system enhanced the PAH oxidation efficiency from 91%
389 (without CPC) to 97% at circumneutral pH (Choi et al., 2014). Recently, use of SDS as
390 surfactant with H₂O₂ + Fe^{III} resulted in higher PAH removal than the systems without SDS
391 including H₂O₂ + Fe^{III} or H₂O₂ + nanoparticles of zerovalent iron (nZVI) (Pardo et al., 2016).
392 It should be noted that surfactants can also be formed as a result of partial oxidation of
393 hydrocarbons and/or native organic matter (Ndjou'ou and Cassidy, 2006; Gryzenia et al.,

394 2009). These authors demonstrated the potential of *in-situ* surfactant production during
395 Fenton oxidation of hydrocarbon-contaminated soils. These surfactants accompanying the
396 Fenton oxidation of hydrocarbons in soil are eventually removed by further chemical
397 oxidation and/or biodegradation (Ndjou'ou and Cassidy, 2006; Gryzenia et al., 2009).
398 Moreover, formation of reductants (e.g. superoxide) during Fenton oxidation could act as
399 surfactant and contribute in enhanced desorption of PAHs (Watts et al., 1999). Non-selective
400 degradation of PAHs was also linked to this probable production of surfactants during
401 Fenton-like oxidation (Usman et al., 2012) that would have rendered HMW PAHs available
402 for remediation (Gryzenia et al., 2009). Pretreatment of contaminated soils with vegetable oil
403 prior to Fenton oxidation also improved the PAH degradation in contaminated soils (Bogan et
404 al., 2003). They incorporated two different oils (corn oil and palm kernel oil) in Fenton
405 treatment for PAH removal from highly contaminated aged soils. A significant increase in the
406 PAH degradation (15–45%) was reported with 5% (per dry wt. of soil) oil addition, with a
407 pronounced effect for HMW PAHs. On the other hand, Fenton treatment (without vegetable
408 oil) could only degrade a few LMW PAHs.

409

410 **2.2.2. Extraction pretreatment**

411 It is well documented that artificially spiked PAHs (fresh contamination) are easier to oxidize
412 than native PAHs because of their binding/sequestration onto soil constituents for years in
413 later case (Nam et al., 2001; Flotron et al., 2005; Usman et al., 2012; Jorfi et al., 2013;
414 Lemaire et al., 2013b). For example, Nam and co-workers (2001) described that Fenton
415 treatment caused higher PAH removal in model spiked soil as compared to that in soil from
416 former manufactured gas plant (MGP). They reported a highly efficient degradation of a
417 mixture of PAHs in the model soil (noticeably, 84.5% and 96.7% of initial pyrene and
418 Benzo(a)pyrene, respectively). In the MGP soil, the same treatment destroyed approximately

419 40% and 20% of pyrene and Benzo(a)pyrene, respectively. Flotron and co-workers (2005)
420 also reported more efficient degradation of spiked PAHs (onto soil, sediment and sludge)
421 than native PAHs in soil after Fenton oxidation.

422 Fenton oxidation ($\text{H}_2\text{O}_2:\text{Fe}^{\text{II}}$ of 20:1) in column experiments under water-unsaturated
423 conditions caused significantly higher PAH degradation for spiked soils (55%) as compared
424 to aged soils (30%) (Lemaire et al., 2013b). In another work, removal of pyrene in freshly
425 contaminated soil was greater (91%) than that of the soil sample with aged pollution (43%)
426 when MF oxidation (300 mM H_2O_2 , 5mM Fe^{III}) using iron nano-particles was applied (Jorfi
427 et al., 2013). The pyrene degradation extent decreased from 88.9% in freshly contaminated
428 soil to 74.7% in soil aged for 30 days although further increase in aging time did not
429 significantly decrease the removal efficiency (Sun and Yan, 2008). As a matter of fact,
430 incubation of a freshly contaminated coking plant soil by a ^{13}C -phenanthrene indicated that
431 bacteria rapidly degraded the freshly-added PAH whereas the native contamination of the
432 coking plant soil (including phenanthrene) is not affected (Cébron et al., 2011). But all these
433 studies using spiked soils were based upon the PAHs whose solutions were freshly prepared
434 in laboratory and were not extracted from the field soil.

435 This was done in a recent study where two ancient coking plant soils contaminated with 16
436 PAHs were extracted and solvent extractable organic matter (EOM) separated from these
437 soils was spiked on sand (Usman et al., 2012). Then, application of Fenton-like oxidation
438 ($\text{H}_2\text{O}_2:\text{Fe}$ of 20:1, 10% magnetite w/w, neutral pH) (FL) resulted in significant degradation of
439 PAHs (90%) in sand spiked with EOM (Fig. 5). It should be noted that FL oxidation was
440 unable to degrade PAHs when applied directly in both soils. It might be correlated to the long
441 term aging of both soils as coking activities and steel industry continued on these soils from
442 the very beginning of the 20th century till 1983 (Ouvrard et al., 2011). As a next step of study,
443 Usman and co-workers (2012) applied extraction pretreatment to re-incorporate the PAHs

444 into the soil as fresh contaminants. It was done by following the same protocol as was used to
445 separate EOM (for spiking sand) but without separating it from the soil, rather it was allowed
446 to evaporate in soil. This procedure allowed PAH removal of 50-60% in re-contaminated
447 soils by successive Fenton-like oxidation (oxidation conditions similar to spiked sand) as
448 indicated in Fig. 5 (Usman et al., 2012). This drop in degradation efficiency between real soil
449 (50-60%) and spiked sand (90%) was correlated to soil matrix effect because of the insoluble
450 organic matter and mineral fraction (Flotron et al., 2005; Jonsson et al., 2007; Usman et al.,
451 2012). But, application of similar pretreatment allowed a six-times higher PAH availability
452 than raw soil measured by Tenax extraction that is closer to bioavailable fraction (Cébron et
453 al., 2013). This difference could be caused by different PAH availability levels including
454 bioavailable (Tenax available) and chemoavailable (deduced by FL oxidation) probably due
455 to higher accessibility for chemical oxidation as suggested elsewhere (Cébron et al., 2013).
456 Very few studies used the extraction pre-treatment to re-incorporate PAHs in same soil as
457 fresh contaminants (Usman et al., 2012; Cébron et al., 2013). Owing to its strong efficiency
458 to improve PAH availability, the *in-situ* feasibility of extraction pretreatment merits to be
459 explored at field scale. It should be noted that cyclodextrin or surfactant-enhanced *in-situ*
460 flushing and extraction of hydrocarbons compounds have been tested at the field scale
461 (McCray and Brusseau, 1999; McCray et al., 2000; Bardi et al., 2007; Chong et al., 2014).
462 However to the best of our knowledge, the use of solubility-enhancement agents in
463 combination with chemical oxidation has been never tested at field scale.

464

465 **2.2.3 Thermal pretreatment**

466 Pre-heating PAH contaminated soils also improved the remediation efficiency of successive
467 Fenton-like treatment (Biache et al., 2015; Usman et al., 2016). Recently, Usman and co-
468 workers (2016) pre-heated the two ancient coking plant soils at different temperatures (60,

469 100, and 150 °C for 1 week under inert atmosphere). Pre-heating resulted in slight removal of
470 PAHs (6% for A soil and 9% for B soil even at 150 °C) but successive FL oxidation
471 (H_2O_2 :Fe of 20:1, 10% magnetite w/w, neutral pH) in pre-heated soils showed significant
472 PAH degradation (19, 29, and 43 % in A soil and 31, 36, and 47 % in B soil pre-treated at 60,
473 100, and 150 °C, respectively, reference for degradation were pre-heated samples) (Fig. 5).
474 No PAH degradation was reported when soils were subjected to chemical oxidation without
475 pre-treatment (Usman et al., 2016). Similarly, Biache and co-workers (2015) reported an
476 increase in PAH degradation in preheated soil samples (100 °C for 1 week under inert
477 atmosphere) by H_2O_2 oxidation systems (including FL oxidation by 14.5 mL of H_2O_2
478 (30%wt), 500 mg magnetite for 5 g soil, neutral pH). Their experiments were based on three
479 kinds of soils that were sampled from former coking plant, former gas plant and an active
480 wood treating facility. Both coking and gas plant soils were characterized by similar EOM
481 (10.3 and 11.5 mg/g, respectively) and PAH content (1146 and 1781 $\mu\text{g/g}$, respectively)
482 while soil from wood-treating facility represented higher amount of EOM (81.94 mg/g) and
483 PAHs (c.a. 12,000 $\mu\text{g.g}^{-1}$). The PAH distribution varied for each sample origin and was
484 dominated by 4- and 5-ring PAHs (coking plant soil) or 2- to 4-ring PAHs (gas plant soil) or
485 by pyrene and fluoranthene (wood treatment facility). Their results indicate that efficiency of
486 FL oxidation varied according to the type of the soil. They did not report any PAH
487 degradation by direct FL oxidation in coking plant soil unless the sample was pre-heated
488 (20% degradation by successive FL oxidation). On the other hand, FL oxidation in the
489 unheated and pre-heated gas plant soils showed the highest degradation rates (78% and 92%,
490 respectively). The oxidative degradation of PAHs reached 49% and 50%, for the unheated
491 and pre-heated wood-treating facility soils, respectively. Results obtained by Biache and co-
492 workers (2015) clearly demonstrate that PAH unavailability was mainly related in coking
493 plant soil due to its long-term aging (Ouvrard et al., 2011; Usman et al., 2012; Usman et al.,

494 2016). It was reported elsewhere that efficiency of Fenton oxidation was negatively
495 correlated to the age of the PAH contamination when experiments were performed by using
496 ten different soil samples originating from former gas plant, former wood-treating facility and
497 active coke production site and characterized by different contamination age ranging from 40
498 to 71 years (Jonsson et al., 2006; Jonsson et al., 2007). In aforementioned coking plant soil
499 (Biache et al. 2015, Usman et al. 2016), no such unavailability constraint was reported during
500 air oxidation of PAHs at 100 °C (Biache et al., 2011) which highlights the role of temperature
501 to control PAH availability. Similarly, heating aged PAH contaminated soils at higher
502 temperature (120 °C) prior to microbial incubation enhanced the PAH degradation due to an
503 increase in their bioavailability (Bonten et al., 1999). Mechanisms governing thermal pre-
504 treatment enhanced PAH availability has not been yet explored due to the limited number of
505 relevant studies. Moreover, further studies are required to test the upscaling of thermal
506 pretreatment for field applications while nature of the soil should be considered.

507

508 **2.3. Deterioration of soil quality and impacts on microbial community**

509 Fenton-based oxidation treatments helps to decrease the PAH load in contaminated soils but
510 it could negatively affect the soil quality and microbial community. This section illustrates
511 the effect of various Fenton-based strategies being used to remove PAHs from contaminated
512 soils.

513 Impact of Fenton oxidation (without pH adjustment or pH adjusted at 3 with H₂SO₄ 1 N) onto
514 soil quality was investigated by Sirguy and co-workers (2008) in spiked agricultural soil and
515 coking plant soil. Significant degradation (75-97%) of PAHs was reported whereas
516 associated decrease in soil pH was found to be the limiting factor for plant growth. This
517 treatment resulted in a significant decline of organic C and N along with strong changes in

518 nutrient pool. Water retention capacity was however improved upon treatment that could
519 encourage re-vegetation (Sirguey et al., 2008).

520 Laurent and co-workers (2012) applied MF oxidation ($\text{H}_2\text{O}_2 + \text{Fe}^{\text{II}}$ without pH adjustment)
521 under unsaturated conditions in industrial PAH-contaminated soil. A degradation of 14% and
522 22% of PAHs was obtained with addition of H_2O_2 at 6 and 65 g kg^{-1} , respectively. The results
523 obtained at higher oxidant dose (65 g kg^{-1}) indicated a significant decrease in soil pH (4.8
524 decreased from 7.3), cation exchange capacity (2 units) and extractable phosphorus content
525 (77% of initial content). Although drop in pH was not as harsh as it could be if initial
526 acidification (low to 3) was adopted, it contributed along with higher oxidant amount to
527 decrease bacterial, fungal, and PAH degrading bacteria population that rebounded after five
528 weeks of incubation (Fig. 6). Plant growth was also negatively affected at higher oxidant
529 dose. Treated soil also displayed a significant delay in seed germination and a decrease in
530 root and shoot biomass (Laurent et al., 2012).

531 A column study was conducted by Palmroth et al. (Palmroth et al., 2006a) where soil pH was
532 not acidified prior to MF oxidation (0.4 $\text{g H}_2\text{O}_2/\text{g}$ soil without any ferrous iron addition) to
533 minimize the stress on soil bacteria. Obtained results indicated a partial removal of PAHs (up
534 to 52%). But toxicity of column leachate was increased towards *Vibrio fischeri* that was used
535 to assess the effect of MF oxidation on microbial activity. Microbial activity and abundance
536 of viable cells was lower in incubated MF-treated soil than in untreated soil. Still, incubation
537 of MF-treated soil slightly improved the PAH-removal than in untreated soil highlighting its
538 compatibility as also reported in another study (Palmroth et al., 2006b).

539 The high concentration of H_2O_2 used and the radicals formed in the Fenton's reaction are
540 toxic to soil bacteria and, therefore, reduce the abundance of viable cells (Büyüksönmez et
541 al., 1998). Apart from that, the acidic pH of 2–3 required for the traditional Fenton oxidation
542 can disturb the soil ecosystem and thus, may be incompatible with biological treatment (Nam

543 et al. 2001). But application of this treatment at near neutral pH could be compatible with
544 bioremediation (Nam et al., 2001). Ecological impacts and environmental concerns due to the
545 presence of H₂O₂ can be significantly addressed by coupling with biological treatment.
546 Moreover, combination of Fenton oxidation and bioremediation was found more efficient to
547 degrade PAHs than sole process and thus, coupling was suggested to improve the removal
548 efficiency and to minimize associated risks (Kulik et al., 2006; Palmroth et al., 2006a;
549 Palmroth et al., 2006b; Valderrama et al., 2009; Venny et al., 2012c). A pilot scale study
550 designed to treat MGP soil revealed that use of Fenton oxidation as the pre-treatment
551 followed by biological treatment using an aerobic biosystem can decrease the PAHs to meet
552 environmental standards (Liu et al., 1993). Moreover, no volatile organic compounds (VOCs)
553 were observed during air monitoring from these experiments implying the safety of this
554 process for environment. Integrated bioremediation and Fenton oxidation in bioslurry
555 reactors also showed more than 80% degradation of PAHs while total microbial counts
556 ranged up to 10⁴ CFU/mL despite the severe oxidation conditions (Brown, 1997).
557 Experimental conditions should be carefully optimized to improve compatibility of both
558 processes as suggested in above studies. Moreover, it should be noted that microbes tend to
559 restore their population and activity after few days (Sahl and Munakata-Marr, 2006; Sirguyey
560 et al., 2008). But, too much H₂O₂ during Fenton oxidation may potentially lead to undesired
561 soil sterilization as it destroys the native micro-biota during the treatment (Valderrama et al.,
562 2009).

563 Venny and co-workers (2012c) demonstrated the strong compatibility of MF oxidation (Fe^{III}
564 + SP as chelating agent) with bioremediation (natural attenuation and biostimulation) for an
565 efficient degradation of PAHs in column experiments simulating in-situ soil flushing. These
566 authors, in another study in packed columns (Venny et al., 2014), described the impact of SP
567 modified Fenton oxidation on soil functionality during remediation of PAHs in spiked and

568 aged soils. Use of optimized operating parameters during MF oxidation ($\text{H}_2\text{O}_2/\text{soil}=0.081$,
569 $\text{Fe}^{\text{III}}/\text{soil}=0.024$, $\text{SP}/\text{soil}=0.024$, pH of SP solution=7.73) could make this treatment
570 compatible with bioremediation and crop growth as the obtained values for pH, electrical
571 conductivity, iron precipitation and soil respiration were within acceptable limits.

572 Gan et al. (Gan et al., 2013) compared the impact of ethyl lectate based Fenton (ELF)
573 treatment onto soil quality with that of conventional water based Fenton treatment (CWF). It
574 was found that ELF caused higher loss of soil organic matter than CWF that was correlated to
575 its high efficiency. Moreover, a decrease in CEC, available P, Cu and Zn was observed in
576 both treatments but to a greater extent in CWF, while ELF caused higher decrease in specific
577 surface area and loss of N. Both treatments, when conducted at low pH (3-4) and high Fe^{II}
578 dosage ($\text{H}_2\text{O}_2:\text{Fe}^{\text{II}} = 10:1$), caused Fe and Al toxicity. But no excessive Fe or Al was formed
579 when ELF was employed at higher pH (≥ 6.2) and lower amounts of added Fe^{II} . This study
580 suggested that ELF adopted for soil with such higher native pH could be compatible for
581 revegetation purposes (Gan et al., 2013).

582 In addition, traditional Fenton oxidation has been known to frequently generate toxic
583 transformation products due to partial mineralization of PAHs that could also contribute
584 towards soil toxicity (Lee and Hosomi, 2001a; Flotron et al., 2005; Lundstedt et al., 2006). It
585 has been reported that anthracene and benzo[a]anthracene were degraded by the Fenton
586 oxidation by 97% and each target substance was further transformed to oxygenated PAHs by
587 68% (Lee and Hosomi, 2001b) and 39% (Lee et al., 1998), respectively. It has been reported
588 that oxy-PAHs are more persistent than the target PAHs and some of them were not degraded
589 any further during Fenton oxidation (Lundstedt et al., 2006). Moreover, owing to the higher
590 solubility, these oxygenated PAHs (dibenzofuran, 9-H fluorenone and anthraquinone) are
591 known to be more mobile than PAHs in US-EPA list (Lundstedt et al., 2014). Thus, their
592 distribution and behavior was also focused along with target PAHs in many chemical

593 oxidation studies (Choi et al., 2014; Biache et al., 2015; Usman et al., 2016). No such by-
594 products were detected during magnetite-catalyzed FL oxidation (Usman et al., 2016) or
595 pyrite-Fenton system (Choi et al., 2014). A similar evolution observed for oxy-PAHs and
596 target PAHs suggested a non-preferential removal by FL oxidation (Usman et al., 2016).

597

598 **3. Conclusions and Outlook**

599 Fenton oxidation is widely used technique to remediate PAHs in contaminated soils but
600 various limitations are associated with its environmental application. This review is intended
601 to provide the readers a critical illustration of various limitations of this process along with a
602 brief discussion of strategies to overcome these limitations. The practical applicability of this
603 method strongly depends on the soil type and structure, history of contamination, and
604 concentration and physical state of the contaminants. A rigorous investigation of all these
605 parameters is a pre-requisite for effective application in soil remediation.

606 Considering the main drawback that its optimal pH is around 3, the traditional Fenton
607 treatment is costly and impractical due to high buffering capacity of soils and associated
608 hazardous effects. Use of modified Fenton oxidation by using chelating agents can allow its
609 use at circumneutral pH but chelating agents can also compete for $\cdot\text{OH}$ radical with PAHs
610 and thus significantly affecting the oxidation efficiency. A recent approach in this regard is
611 the use of inorganic chelating agents that offers advantages over organic chelating agents for
612 oxidant consumption. Another possibility is the use of iron minerals, which are able to
613 catalyze Fenton-like oxidation over a wide range of pH. Due to the presence of structural Fe^{II} ,
614 magnetite showed strong reactivity to catalyze Fenton-like oxidation of PAHs at
615 circumneutral pH. Moreover, its catalytic stability can be maintained for multiple oxidation
616 cycles and it can be a promising treatment. Use of iron minerals or nanoparticles of ZVI
617 could provide a cost-effective alternative to the use of more than one reagent (e.g. chelating

618 agent + soluble Fe^{II}). Further studies are needed for a better understanding of Fenton-like
619 application in real field conditions, as injection of iron particles in unsaturated zone such as
620 vadose zone requires many investigations. Mobility, transport and fate of such particles
621 should be addressed on context of reactivity, stability and eco-toxicity. Detailed studies are
622 required to exploit the potential of endogenous iron minerals to catalyze chemical oxidation.
623 Another major limitation is the PAH unavailability during Fenton oxidation. Many strategies
624 are discussed including the use of availability enhancement agents, extraction or thermal
625 pretreatment. PAH availability is the crucial factor controlling its degradation in aged soils
626 and that should be considered while designing a remediation treatment. Use of suitable
627 pretreatment to enhance PAH availability can enhance the treatment efficiency. Most of the
628 studies are based on the batch experiments that should be up-scaled to column experiments
629 under saturated and unsaturated conditions to get closer to real field conditions. The field
630 application of certain pretreatments (extraction or thermal) seems difficult to implement that
631 merits to be resolved but they can serve as good approach for *ex-situ* application. Integration
632 of Fenton's treatment with bioremediation could become a suitable option to address the
633 detrimental effects associated with Fenton's application. The development of new modeling
634 approaches that account for the up-scaling of different involved reactions and complexity of
635 soil system becomes urgent in soil remediation science. More experimental work is also
636 needed to develop strategies with high remediation efficiency and yet are suitable for soil
637 quality so that contaminated sites can be restored to productive use.

638

639 **Acknowledgment**

640 This work was supported by a post-doctoral research fellowship from the Alexander von
641 Humboldt Foundation, which is gratefully acknowledged.

642

643

644
645
646
647
648
649
650
651
652
653
654
655
656
657
658
659
660
661
662
663
664
665
666
667
668
669
670
671
672
673
674
675
676
677
678
679
680
681
682
683
684
685
686
687
688
689
690
691
692

References

- Bardi L, Mattei A, Steffan S, Marzona M. Hydrocarbon degradation by a soil microbial population with [beta]-cyclodextrin as surfactant to enhance bioavailability. *Enzyme Microb. Technol.* 2000; 27: 709-713.
- Bardi L, Martini C, Opsi F, Bertolone E, Belviso S, Masoero G, Marzona M, Marsan FA. Cyclodextrin-enhanced in situ bioremediation of polyaromatic hydrocarbons-contaminated soils and plant uptake. *J. Incl. Phenom. Macro. Chem.* 2007; 57: 439-444.
- Biache C, Ghislain T, Faure P, Mansuy-Huault L. Low temperature oxidation of a coking plant soil organic matter and its major constituents: An experimental approach to simulate a long term evolution. *J. Hazard. Mater.* 2011; 188: 221-230.
- Biache C, Lorgeoux C, Andriatsihoarana S, Colombano S, Faure P. Effect of pre-heating on the chemical oxidation efficiency: Implications for the PAH availability measurement in contaminated soils. *J. Hazard. Mater.* 2015; 286: 55-63.
- Bogan BW, Trbovic V, Paterek JR. Inclusion of vegetable oils in Fenton's chemistry for remediation of PAH-contaminated soils. *Chemosphere* 2003; 50: 15-21.
- Bonten LTC, Grotenbuis TC, Rulkens WH. Enhancement of PAH biodegradation in soil by physicochemical pretreatment. *Chemosphere* 1999; 38: 3627-3636.
- Brown K. Innovative methods for bioslurry treatment. Site Remediation Technology Report. National Risk Management Research Laboratory. Office of Research and Development. US Environment Protection Agency, Cincinnati, Ohio 1997; 45268.
- Büyüksönmez F, Hess TF, Crawford RL, Watts RJ. Toxic effects of modified Fenton reactions on *Xanthobacter flavus* FB71. *Appl. Environ. Microb.* 1998; 64: 3759-3764.
- Catrouillet C, Davranche M, Dia A, Bouhnik-Le Coz M, Marsac R, Pourret O, Gruau G. Geochemical modeling of Fe(II) binding to humic and fulvic acids. *Chem. Geol.* 2014; 372: 109-118.
- Cébron A, Louvel B, Faure P, France-Lanord C, Chen Y, Murrell JC, Leyval C. Root exudates modify bacterial diversity of phenanthrene degraders in PAH-polluted soil but not phenanthrene degradation rates. *Environ. Microbiol.* 2011; 13: 722-736.
- Cébron A, Faure P, Lorgeoux C, Ouvrard S, Leyval C. Experimental increase in availability of a PAH complex organic contamination from an aged contaminated soil: Consequences on biodegradation. *Environ. Pollut.* 2013; 177: 98-105.
- Choi K, Bae S, Lee W. Degradation of pyrene in cetylpyridinium chloride-aided soil washing wastewater by pyrite Fenton reaction. *Chem. Eng. J.* 2014; 249: 34-41.
- Chong Z-Y, Liao X-Y, Yan X-L, Sun L, Zhao D, Liang T. Enhanced Desorption of PAHs from Manufactured Gas Plant Soils Using Different Types of Surfactants. *Pedosphere* 2014; 24: 209-219.
- Cornell RM, Schwertmann U. *The Iron Oxides: Structure, Properties, Reactions, Occurrence and Uses*: Wiley-VCH, 2006.
- E.E.A. <http://www.eea.europa.eu/data-and-maps/figures/overview-ofcontaminants-affecting-soil-and-groundwater-in-europe>, 2012.
- Ershadi L, Ebadi T, Ershadi V, Rabbani AR. Chemical oxidation of crude oil in oil contaminated soil by Fenton process using nano zero valent Iron. 2nd International Conference on Environmental Science and Technology. 6, 2011, pp. V1-89.
- Fenton HJH. Oxidation of tartaric acid in presence of iron. *J. Chem. Soc. T.* 1894; 65: 899-910.
- Ferrarese E, Andreottola G, Oprea IA. Remediation of PAH-contaminated sediments by chemical oxidation. *J. Hazard. Mater.* 2008; 152: 128-139.

693 Flotron V, Delteil C, Padellec Y, Camel V. Removal of sorbed polycyclic aromatic
694 hydrocarbons from soil, sludge and sediment samples using the Fenton's reagent
695 process. *Chemosphere* 2005; 59: 1427-1437.

696 Gallard H, de Laat J, Legube B. Effect of pH on the oxidation rate of organic compounds by
697 Fe-II/H₂O₂. Mechanisms and simulation. *New J. Chem.* 1998; 22: 263-268.

698 Gan S, Lau EV, Ng HK. Remediation of soils contaminated with polycyclic aromatic
699 hydrocarbons (PAHs). *J. Hazard. Mater.* 2009; 172: 532-549.

700 Gan S, Yap CL, Ng HK, Venny. Investigation of the impacts of ethyl lactate based Fenton
701 treatment on soil quality for polycyclic aromatic hydrocarbons (PAHs)-contaminated
702 soils. *J. Hazard. Mater.* 2013; 262: 691-700.

703 Garrido-Ramírez EG, Theng BKG, Mora ML. Clays and oxide minerals as catalysts and
704 nanocatalysts in Fenton-like reactions — A review. *Appl. Clay Sci.* 2010; 47: 182-
705 192.

706 Georgi A, Schierz A, Trommler U, Horwitz CP, Collins TJ, Kopinke FD. Humic acid
707 modified Fenton reagent for enhancement of the working pH range. *Appl. Catal. B,*
708 *Environ.* 2007; 72: 26-36.

709 Gryzenia J, Cassidy D, Hampton D. Production and accumulation of surfactants during the
710 chemical oxidation of PAH in soil. *Chemosphere* 2009; 77: 540-545.

711 Hanna K, Chiron S, Oturan MA. Coupling enhanced water solubilization with cyclodextrin to
712 indirect electrochemical treatment for pentachlorophenol contaminated soil
713 remediation. *Water Res.* 2005; 39: 2763-2773.

714 Hanna K, Kone T, Medjahdi G. Synthesis of the mixed oxides of iron and quartz and their
715 catalytic activities for the Fenton-like oxidation. *Catal. Commun.* 2008; 9: 955-959.

716 Hanna K, Usman M, Chatain V. Sorption and transport of aqueous FeII in a goethite-coated
717 sand column under anoxic conditions. *Appl. Geochem.* 2013.

718 Hannah W, Thompson PB. Nanotechnology, risk and the environment: a review. *J. Environ.*
719 *Monit.* 2008; 10: 291-300.

720 Haritash AK, Kaushik CP. Biodegradation aspects of Polycyclic Aromatic Hydrocarbons
721 (PAHs): A review. *J. Hazard. Mater.* 2009; 169: 1-15.

722 ITRC. Technical and Regulatory Guidance for In Situ Chemical Oxidation of Contaminated
723 Soil and Groundwater. ITRC, ISCO Team, Washington, DC, 2005.

724 Jiang X, Tong M, Li H, Yang K. Deposition kinetics of zinc oxide nanoparticles on natural
725 organic matter coated silica surfaces. *J. Colloid Interface Sci.* 2010; 350: 427-434.

726 Johnson WP, Amy GL. Facilitated Transport and Enhanced Desorption of Polycyclic
727 Aromatic Hydrocarbons by Natural Organic Matter in Aquifer Sediments. *Environ.*
728 *Sci. Technol.* 1995; 29: 807-817.

729 Jonsson S, Persson Y, Frankki S, Lundstedt S, van Bavel B, Haglund P, Tysklind M.
730 Comparison of Fenton's reagent and ozone oxidation of polycyclic aromatic
731 hydrocarbons in aged contaminated soils. *J. Soils Sediment.* 2006; 6: 208-214.

732 Jonsson S, Persson Y, Frankki S, van Bavel B, Lundstedt S, Haglund P, Tysklind M.
733 Degradation of polycyclic aromatic hydrocarbons (PAHs) in contaminated soils by
734 Fenton's reagent: A multivariate evaluation of the importance of soil characteristics
735 and PAH properties. *J. Hazard. Mater.* 2007; 149: 86-96.

736 Jorfi S, Rezaee A, Moheb-ali G-a, Jaafarzadeh Na. Pyrene removal from contaminated soils
737 by modified Fenton oxidation using iron nano particles. *J. Environ. Health Sci. Eng.*
738 2013; 11: 17.

739 Jung YS, Lim WT, Park JY, Kim YH. Effect of pH on Fenton and Fenton- like oxidation.
740 *Environ. Technol.* 2009; 30: 183-190.

741 Kanel SR, Neppolian B, Choi H, Yang JW. Heterogeneous catalytic oxidation of
742 phenanthrene by hydrogen peroxide in soil slurry: Kinetics, mechanism, and
743 implication. *Soil Sediment Contam.* 2003; 12: 101-117.

744 Kanel SR, Neppolian B, Jung H, Choi H. Comparative removal of polycyclic aromatic
745 hydrocarbons using iron oxide and hydrogen peroxide in soil slurries. *Environ. Eng.*
746 *Sci.* 2004; 21: 741-751.

747 Kong SH, Watts RJ, Choi JH. Treatment of petroleum-contaminated soils using iron mineral
748 catalyzed hydrogen peroxide. *Chemosphere* 1998; 37: 1473-1482.

749 Kulik N, Goi A, Trapido M, Tuhkanen T. Degradation of polycyclic aromatic hydrocarbons
750 by combined chemical pre-oxidation and bioremediation in creosote contaminated
751 soil. *J. Environ. Manag.* 2006; 78: 382-391.

752 Laurent F, Cébron A, Schwartz C, Leyval C. Oxidation of a PAH polluted soil using
753 modified Fenton reaction in unsaturated condition affects biological and physico-
754 chemical properties. *Chemosphere* 2012; 86: 659-664.

755 Lee BD, Hosomi M, Murakami A. Fenton oxidation with ethanol to degrade anthracene into
756 biodegradable 9, 10-anthraquinone: A pretreatment method for anthracene-
757 contaminated soil. *Water Sci. Technol.* 1998; 38: 91-97.

758 Lee BD, Hosomi M. A hybrid Fenton oxidation-microbial treatment for soil highly
759 contaminated with benz(a)anthracene. *Chemosphere* 2001a; 43: 1127-1132.

760 Lee BD, Hosomi M. Fenton oxidation of ethanol-washed distillation-concentrated
761 benzo(A)pyrene: Reaction product identification and biodegradability. *Water Res.*
762 2001b; 35: 2314-2319.

763 Lee BD, Nakai S, Hosomi M. Application of fenton oxidation to remediate polycyclic
764 aromatic hydrocarbons-contaminated soil. *J. Chem. Eng. Jpn.* 2002; 35: 582-586.

765 Lemaire J, Buès M, Kabeche T, Hanna K, Simonnot M-O. Oxidant selection to treat an aged
766 PAH contaminated soil by in situ chemical oxidation. *J. Environ. Chem. Eng.* 2013a;
767 1: 1261-1268.

768 Lemaire J, Laurent F, Leyval C, Schwartz C, Buès M, Simonnot M-O. PAH oxidation in
769 aged and spiked soils investigated by column experiments. *Chemosphere* 2013b; 91:
770 406-414.

771 Lindsey ME, Xu G, Lu J, Tarr MA. Enhanced Fenton degradation of hydrophobic organics
772 by simultaneous iron and pollutant complexation with cyclodextrins. *Sci. Total*
773 *Environ.* 2003; 307: 215-229.

774 Lingle J, Hellman E, Noel M, Wilson J. Full Scale Chemical Oxidation of Manufactured Gas
775 Plant Hydrocarbons in Soil and Groundwater Using Fenton's Reagent, 2001.

776 Liu BY, Srivastava V, Paterek J, Pradhan S, Pope J, Hayes T, Linz D, Jerger D. MGP soil
777 remediation in a slurry-phase system: a pilot-scale test. Institute of Gas Technology,
778 Chicago, IL (United States), 1993.

779 Lowry GV, Gregory KB, Apte SC, Lead JR. Transformations of Nanomaterials in the
780 Environment. *Environ. Sci. Technol.* 2012; 46: 6893-6899.

781 Lundstedt S, Persson Y, Öberg L. Transformation of PAHs during ethanol-Fenton treatment
782 of an aged gasworks' soil. *Chemosphere* 2006; 65: 1288-1294.

783 Lundstedt S, Bandowe BAM, Wilcke W, Boll E, Christensen JH, Vila J, Grifoll M, Faure P,
784 Biache C, Lorgeoux C. First intercomparison study on the analysis of oxygenated
785 polycyclic aromatic hydrocarbons (oxy-PAHs) and nitrogen heterocyclic polycyclic
786 aromatic compounds (N-PACs) in contaminated soil. *TrAC, Trends Anal. Chem.*
787 2014; 57: 83-92.

788 Martens DA, Frankenberger Jr WT. Enhanced degradation of polycyclic aromatic
789 hydrocarbons in soil treated with an advanced oxidative process—Fenton's Reagent.
790 *Soil Sediment Contam.* 1995; 4: 175-190.

791 Matta R, Hanna K, Chiron S. Fenton-like oxidation of 2,4,6-trinitrotoluene using different
792 iron minerals. *Sci. Total Environ.* 2007; 385: 242-251.

793 McCray JE, Brusseau ML. Cyclodextrin-Enhanced In Situ Flushing of Multiple-Component
794 Immiscible Organic Liquid Contamination at the Field Scale: Analysis of Dissolution
795 Behavior. *Environ. Sci. Technol.* 1999; 33: 89-95.

796 McCray JE, Boving TB, Brusseau ML. Cyclodextrin-Enhanced Solubilization of Organic
797 Contaminants with Implications for Aquifer Remediation. *Ground Water Monit. R.*
798 2000; 20: 94-103.

799 Nam K, Rodriguez W, Kukor JJ. Enhanced degradation of polycyclic aromatic hydrocarbons
800 by biodegradation combined with a modified Fenton reaction. *Chemosphere* 2001; 45:
801 11-20.

802 Ndjou'ou AC, Cassidy D. Surfactant production accompanying the modified Fenton
803 oxidation of hydrocarbons in soil. *Chemosphere* 2006; 65: 1610-1615.

804 Ouvrard S, Barnier C, Bauda P, Beguiristain T, Biache C, Bonnard M, Caupert C, Cébron A,
805 Cortet J, Cotellet S, Dazy M, Faure P, Masfarau JF, Nahmani J, Palais F, Poupin P,
806 Raoult N, Vasseur P, Morel JL, Leyval C. In Situ Assessment of Phytotechnologies
807 for Multicontaminated Soil Management. *Int. J. Phytorem.* 2011; 13: 245-263.

808 Palmroth MRT, Langwaldt JH, Aunola TA, Goi A, Münster U, Puhakka JA, Tuhkanen TA.
809 Effect of modified Fenton's reaction on microbial activity and removal of PAHs in
810 creosote oil contaminated soil. *Biodegradation* 2006a; 17: 131-141.

811 Palmroth MRT, Langwaldt JH, Aunola TA, Goi A, Puhakka JA, Tuhkanen TA. Treatment of
812 PAH-contaminated soil by combination of Fenton's reaction and biodegradation. *J.*
813 *Chem. Technol. Biotechnol.* 2006b; 81: 598-607.

814 Pardo F, Rosas J, Santos A, Romero A. Remediation of a biodiesel blend-contaminated soil
815 by using a modified Fenton process. *Environ Sci Pollut Res* 2014; 21: 12198-12207.

816 Pardo F, Peluffo M, Santos A, Romero A. Optimization of the application of the Fenton
817 chemistry for the remediation of a contaminated soil with polycyclic aromatic
818 hydrocarbons. *J. Chem. Technol. Biotechnol.* 2016; 91: 1763-1772.

819 Park J-Y, Kim J-H. Role of sol with iron oxyhydroxide/sodium dodecyl sulfate composites
820 on Fenton oxidation of sorbed phenanthrene in sand. *J. Environ. Manag.* 2013; 126:
821 72-78.

822 Pignatello JJ, Liu D, Huston P. Evidence for an Additional Oxidant in the Photoassisted
823 Fenton Reaction. *Environ. Sci. Technol.* 1999; 33: 1832-1839.

824 Pignatello JJ, Oliveros E, MacKay A. Advanced oxidation processes for organic contaminant
825 destruction based on the fenton reaction and related chemistry. *Crit. Rev. Environ.*
826 *Sci. Technol.* 2006; 36: 1-84.

827 Qiang Z, Chang JH, Huang CP, Cha D. Oxidation of Selected Polycyclic Aromatic
828 Hydrocarbons by the Fenton's Reagent: Effect of Major Factors Including Organic
829 Solvent. *Nuclear Site Remediation.* 778. American Chemical Society, 2000, pp. 187-
830 209.

831 Rastogi A, Al-Abed SR, Dionysiou DD. Effect of inorganic, synthetic and naturally
832 occurring chelating agents on Fe(II) mediated advanced oxidation of chlorophenols.
833 *Water Res.* 2009; 43: 684-694.

834 Riding MJ, Doick KJ, Martin FL, Jones KC, Semple KT. Chemical measures of
835 bioavailability/bioaccessibility of PAHs in soil: Fundamentals to application. *J.*
836 *Hazard. Mater.* 2013; 261: 687-700.

837 Rivas FJ. Polycyclic aromatic hydrocarbons sorbed on soils: A short review of chemical
838 oxidation based treatments. *J. Hazard. Mater.* 2006; 138: 234-251.

839 Safarzadeh-Amiri A, Bolton JR, Cater SR. The use of iron in advanced oxidation processes.
840 *J. Adv. Oxid. Technol.* 1996: 18-26.

841 Sahl J, Munakata-Marr J. The effects of in situ chemical oxidation on microbiological
842 processes: A review. *Remed. J.* 2006; 16: 57-70.

843 Sillanpaa M, Pirkanniemi K. Recent developments in chelate degradation. *Environ. Technol.*
844 2001; 22: 791-801.

845 Silva PTdSe, Silva VLd, Neto BdB, Simonnot MO. Phenanthrene and pyrene oxidation in
846 contaminated soils using Fenton's reagent. *J. Hazard. Mater.* 2009; 161: 967-973.

847 Sirguy C, Tereza de Souza e Silva P, Schwartz C, Simonnot M-O. Impact of chemical
848 oxidation on soil quality. *Chemosphere* 2008; 72: 282-289.

849 Sun H-w, Yan Q-s. Influence of pyrene combination state in soils on its treatment efficiency
850 by Fenton oxidation. *J. Environ. Manag.* 2008; 88: 556-563.

851 Tamura H, Kawamura S, Hagayama M. Acceleration of the oxidation of Fe²⁺ ions by
852 Fe(III)-oxyhydroxides. *Corros. Sci.* 1980; 20: 963-971.

853 Taylor KM. The Chemical Oxidation of Polycyclic Aromatic Hydrocarbons at a Former
854 Manufactured Gas Plant in Bay Shore, New York. 2012.

855 Usman M, Faure P, Ruby C, Hanna K. Remediation of PAH-contaminated soils by magnetite
856 catalyzed Fenton-like oxidation. *Appl. Catal. B, Environ.* 2012; 117-118: 10-17.

857 Usman M, Faure P, Lorgeoux C, Ruby C, Hanna K. Treatment of hydrocarbon contamination
858 under flow through conditions by using magnetite catalyzed chemical oxidation.
859 *Environ. Sci. Pollut. Res.* 2013; 20: 22-30.

860 Usman M, Chaudhary A, Biache C, Faure P, Hanna K. Effect of thermal pre-treatment on the
861 availability of PAHs for successive chemical oxidation in contaminated soils.
862 *Environ. Sci. Pollut. Res.* 2016; 23: 1371-1380.

863 Valderrama C, Alessandri R, Aunola T, Cortina JL, Gamisans X, Tuhkanen T. Oxidation by
864 Fenton's reagent combined with biological treatment applied to a creosote-
865 contaminated soil. *J. Hazard. Mater.* 2009; 166: 594-602.

866 Veignie E, Rafin C, Landy D, Fourmentin S, Surpateanu G. Fenton degradation assisted by
867 cyclodextrins of a high molecular weight polycyclic aromatic hydrocarbon
868 benzo[a]pyrene. *J. Hazard. Mater.* 2009; 168: 1296-1301.

869 Venny, Gan S, Ng HK. Current status and prospects of Fenton oxidation for the
870 decontamination of persistent organic pollutants (POPs) in soils. *Chem. Eng. J.*
871 2012a; 213: 295-317.

872 Venny, Gan S, Ng HK. Inorganic chelated modified-Fenton treatment of polycyclic aromatic
873 hydrocarbon (PAH)-contaminated soils. *Chem. Eng. J.* 2012b; 180: 1-8.

874 Venny, Gan S, Ng HK. Modified Fenton oxidation of polycyclic aromatic hydrocarbon
875 (PAH)-contaminated soils and the potential of bioremediation as post-treatment. *Sci.*
876 *Total Environ.* 2012c; 419: 240-249.

877 Venny, Gan S, Ng H. Evaluation of in situ catalysed hydrogen peroxide propagation (CHP)
878 for phenanthrene and fluoranthene removals from soil and its associated impacts on
879 soil functionality. *Environ. Sci. Pollut. Res.* 2014; 21: 2888-2897.

880 Viglianti C, Hanna K, de Brauer C, Germain P. Removal of polycyclic aromatic
881 hydrocarbons from aged-contaminated soil using cyclodextrins: Experimental study.
882 *Environ. Pollut.* 2006; 140: 427-435.

883 Watts RJ, Bottenberg BC, Hess TF, Jensen MD, Teel AL. Role of reductants in the enhanced
884 desorption and transformation of chloroaliphatic compounds by modified Fenton's
885 reactions. *Environ. Sci. Technol.* 1999; 33: 3432-3437.

886 Watts RJ, Stanton PC, Howsawkung J, Teel AL. Mineralization of a sorbed polycyclic
887 aromatic hydrocarbon in two soils using catalyzed hydrogen peroxide. *Water Res.*
888 2002; 36: 4283-4292.

- 889 Xue X, Hanna K, Abdelmoula M, Deng N. Adsorption and oxidation of PCP on the surface
890 of magnetite: Kinetic experiments and spectroscopic investigations. *Appl. Catal. B,*
891 *Environ.* 2009a; 89: 432-440.
- 892 Xue X, Hanna K, Despas C, Wu F, Deng N. Effect of chelating agent on the oxidation rate of
893 PCP in the magnetite/H₂O₂ system at neutral pH. *J. Mol. Catal. A: Chem.* 2009b;
894 311: 29-35.
- 895 Yang Y, Ratté D, Smets BF, Pignatello JJ, Grasso D. Mobilization of soil organic matter by
896 complexing agents and implications for polycyclic aromatic hydrocarbon desorption.
897 *Chemosphere* 2001; 43: 1013-1021.
- 898 Yap C, Gan S, Ng H. Feasibility of treating aged polycyclic aromatic hydrocarbons (PAHs)-
899 contaminated soils using ethyl lactate-based Fenton treatment via parametric and
900 kinetic studies. *Environ. Sci. Pollut. Res.* 2015; 22: 329-342.
- 901 Yap CL, Gan S, Ng HK. Fenton based remediation of polycyclic aromatic hydrocarbons-
902 contaminated soils. *Chemosphere* 2011; 83: 1414-1430.
- 903 Yap CL, Gan S, Ng HK. Ethyl lactate-Fenton treatment of soil highly contaminated with
904 polycyclic aromatic hydrocarbons (PAHs). *Chem. Eng. J.* 2012; 200–202: 247-256.

905

906

907 **Figure captions**

908 **Figure 1:** Different phases of experimental setup used in laboratory to investigate application
909 of Fenton oxidation to remediate PAHs in contaminated soils.

910 **Figure 2. (a)** Removal of phenanthrene (PHE) and fluoranthene (FLUT) by using modified-
911 Fenton oxidation in the presence of inorganic chelating agent- SP (sodium pyrophosphate)
912 and organic chelating agents including EDTA (ethylenediaminetetraacetic acid), OA (oxalic
913 acid), SC (sodium citrate) and MA (malic acid). **(b)** H₂O₂ residual for MF treatment in the
914 presence of inorganic (SP) and organic chelating agents (EDTA, OA, SC and MA). Adapted
915 with permission from Figure 2 and Figure 5 of Venny et al. (2012b).

916 **Figure 3:** PAHs degradation in (a) organic extracts (spiked on sand) and (b) soil pretreated
917 with availability enhancement agent during oxidation experiments by: H₂O₂ alone without
918 iron activation (■-HP), H₂O₂ + soluble Fe^{II} (○-F) and H₂O₂ + magnetite (●-FL). Blank (▲)
919 experiment was performed in the presence of magnetite but without any oxidant. This
920 degradation is represented in terms of C_t/C₀ where C_t is the sum of 16 PAHs concentration at
921 specified oxidation time and C₀ is their concentration at t=0 (before oxidation) measured by
922 GC-MS. Adapted with permission from Fig. 2 and Fig. 6 of Usman et al. (2012).
923 Experimental conditions were: solid matrix = 2 g, volume of solution = 20 mL, magnetite
924 (10%, w/w) and equivalent amount of Fe^{II}.

925 **Figure 4:** Efficiency of ethanol in comparison with other PAH-availability enhancement
926 agent to assist Fenton oxidation where F: traditional Fenton, ETF: ethanol-Fenton system,
927 ELF = ethyl lactate-Fenton system, SDS-F = sodium dodecyl sulfate- Fenton system, CD-F =
928 cyclodextrin-Fenton system. Instead of traditional Fenton oxidation, Usman and co-workers
929 (2012) applied magnetite catalyzed Fenton-like oxidation (FL) assisted by ET and CD.

930

931 **Figure 5:** Degradation of 16 PAHs in former coking plant soil by Fenton-like oxidation
932 assisted by various pretreatments and is represented by C/C_0 where C is the final
933 concentration of PAHs after treatment and C_0 is the initial concentration. Where initial =
934 PAH concentration before treatment and others represent application of Fenton-like oxidation
935 direct (Direct oxidation = when soils were subjected to chemical oxidation without any pre-
936 treatment), spiking/FL (when EOM was extracted from soils and spiked on sand), Ext-Pt/FL
937 (extraction pretreatment to incorporate PAHs as fresh contaminants in soil as detailed in
938 Section 2.2.2), and FL oxidation assisted by ethanol (ET-FL) or cyclodextrin (CD) or by
939 thermal pretreatment at 60, 100, and 150 °C.

940 **Figure 6.** Density of cultivable bacteria, 16S rDNA, PAH-RDH α and 18s rDNA gene copy
941 numbers in oxidized by Fenton treatment at low (L) i.e. 6 g kg⁻¹ and high (H) i.e. 65 g kg⁻¹
942 concentration of H₂O₂ and control soil (Control) at the start (T_0) and after 5 weeks (T_5) of
943 incubation. As the evolution of PAH-RHD α GN and GP populations was not significantly
944 different, the two populations were added to represent the global population of degraders.
945 Mean \pm SE ($n = 3$). LD: detection limit. Different letters within category data indicate
946 significant differences between treatments; stars indicate significant differences
947 between T_0 and T_6 by Mann and Whitney test. Reproduced with permission from Fig. 2 of
948 Laurent et al. (2012).

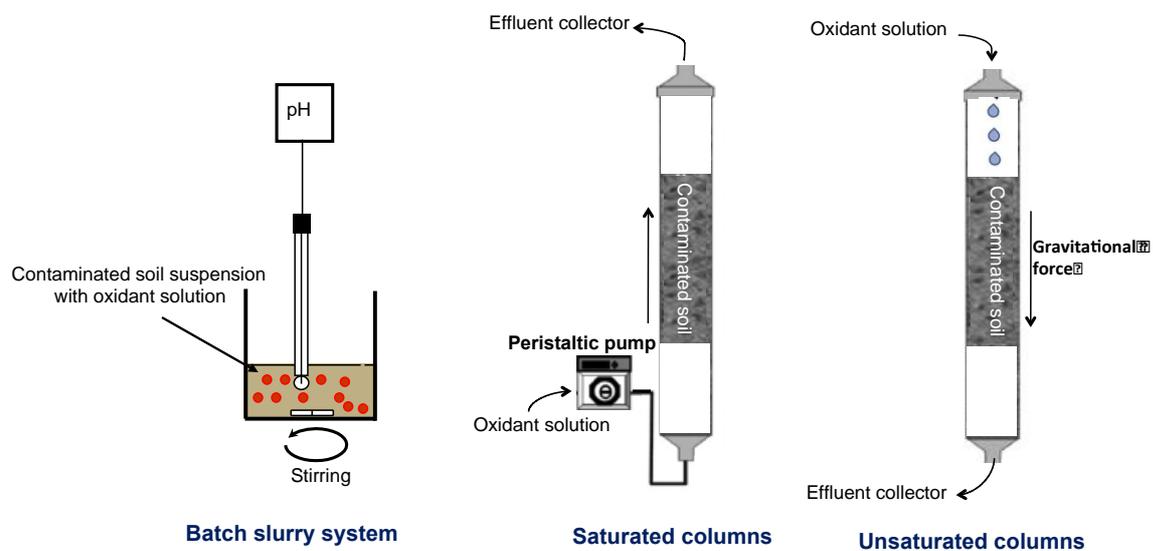
949

950 **Table 1:** Research studies using Fenton oxidation assisted by various kinds of chelating agents.

| Focus of study | Soil | Tested chelating agents (CA) | Experimental conditions | Important results | Reference |
|--|--|--|--|--|-----------------------|
| To compare the efficiency of traditional Fenton and modified Fenton to couple with biodegradation for enhanced PAH removal | Spiked soil (PAHs = 600 mg/Kg), silt loam and soil sampled from former manufactured gas plant site, loamy sand | Catechol (CA) and gallic acid (GA) | 5 g of MGP soil in 20 ml of distilled water, Fe ^{III} ; 82.5 mg of catechol or 141.0 mg of gallic acid and Fe(ClO ₄) ₃ ·6H ₂ O (0.07 M), 2 g of H ₂ O ₂ , neutral pH For traditional Fenton, experiments were performed at pH = 2-3 without chelating agents | Traditional Fenton was effective but required pH (pH 2–3) for optimum efficiency made the process incompatible with biological treatment. Efficiency was maintained by using chelating agents that allowed coupling with bioremediation. | Nam et al., 2001 |
| Production of surfactants during various chemical oxidation strategies for PAH remediation | Soil sampled from former manufactured gas plant site, having sand:silt:clay =79:20:1% | EDTA | Laboratory slurry reactors, 4 kg of soil (dry weight) and 8.5 L of water (including the oxidant solutions), 200 mg of EDTA, 1 g Fe ^{III} (as ferric sulfate), and 1 L of 50% H ₂ O ₂ , pH=8 | All tested treatments resulted significant surfactant production and overall removal of PAHs. Modified Fenton with calcium peroxide exhibited better efficiency. | Gryzenia et al., 2009 |
| Effect of various chelating agents on the efficiency of Fenton oxidation to remove PAHs | Spiked soil (loamy sand), containing 1000 mg/kg of two PAHs | Sodium pyrophosphate (SP) as organic CA, EDTA, Oxalic acid (OA), | Batch slurry system, 5 g of soil, 480 mM H ₂ O ₂ , 60 mM Fe ³⁺ , solution:soil = 15 mL:5 g. Different doses of CAs: 0.1135–0.4015 g | Inorganic SP was demonstrated to be superior in efficiency to the tested organic CAs. | Venny et al., 2012c |

| | | | | | |
|--|---|---|--|---|---------------------|
| | | sodium citrate (SC) as inorganic CAs | | | |
| Optimization of experimental conditions for efficient coupling of Fenton's reagent with bioremediation | Soil (loamy sand) spiked with two PAHs, 500 mg the PAH/kg for each PAH, | SP | Unsaturated soil columns, 1.2 Kg soil, Various doses of oxidant, SP, Fe ^{III} at different reaction times with soil pH of 4.07–5.41 | An optimized operating condition of the modified Fenton was found at H ₂ O ₂ /soil 0.05, Fe ^{III} /soil 0.025, SP/soil 0.04 and 3 h reaction time with 79.42% and 68.08% removal of tested PAHs. Use of SP assisted Fenton treatment and bioremediation serves as a suitable strategy to enhance soil quality. | Venny et al., 2012b |
| Role of various chelating agents to improve operating range of pH for degradation of PAHs by modified Fenton oxidation using iron nano particles | Spiked soil (silty sand), containing 100 mg of pyrene per kg of soil | SP as inorganic CA and EDTA, FA, HA and SC (as organic CAs) | Batch experiments, 5 g soil, 15 mL distilled water, iron nano-oxide = 5 – 60 mM, H ₂ O ₂ = 0-500 mM, pH = 3 and 7 | Application of modified Fenton treatment, with the following conditions (300 mM of H ₂ O ₂ , 30 mM of iron nano oxide, SP as CA, pH 7 after 6 h of reaction, removed 93% of tested PAHs that was higher than organic CAs. | Jorfi et al., 2013 |

952 **Figure 1:**

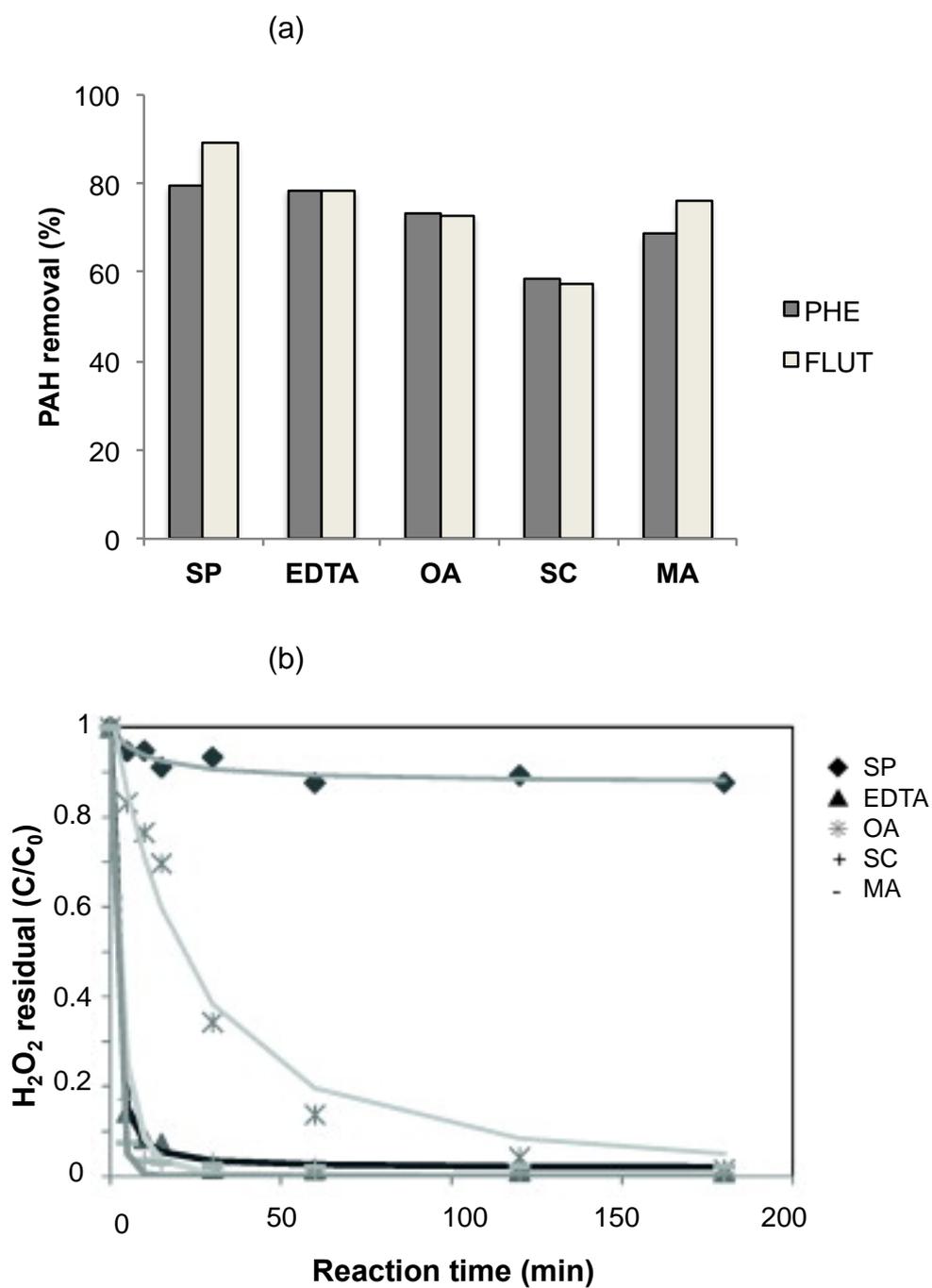


953

954

955 **Figure 2:**

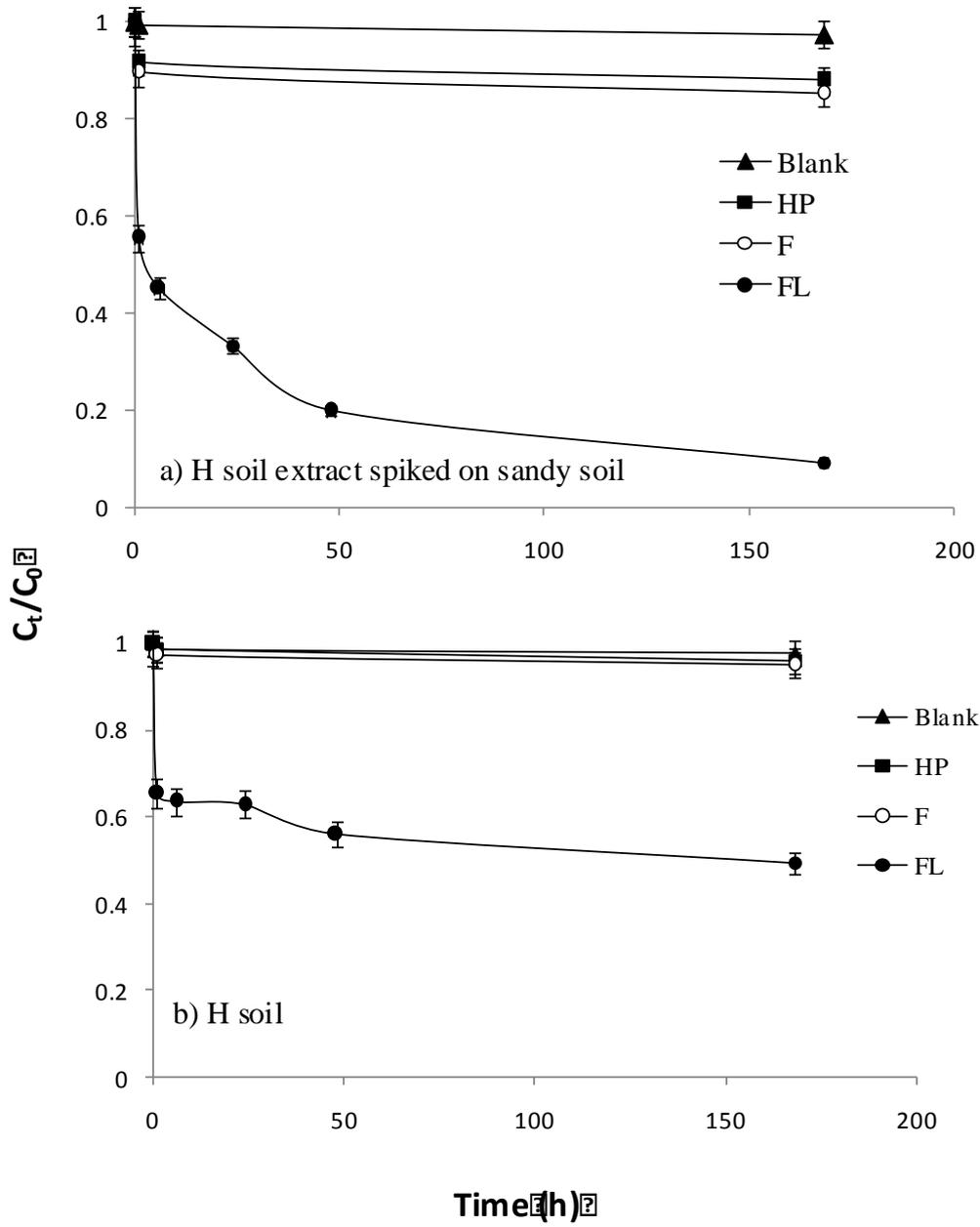
956



957

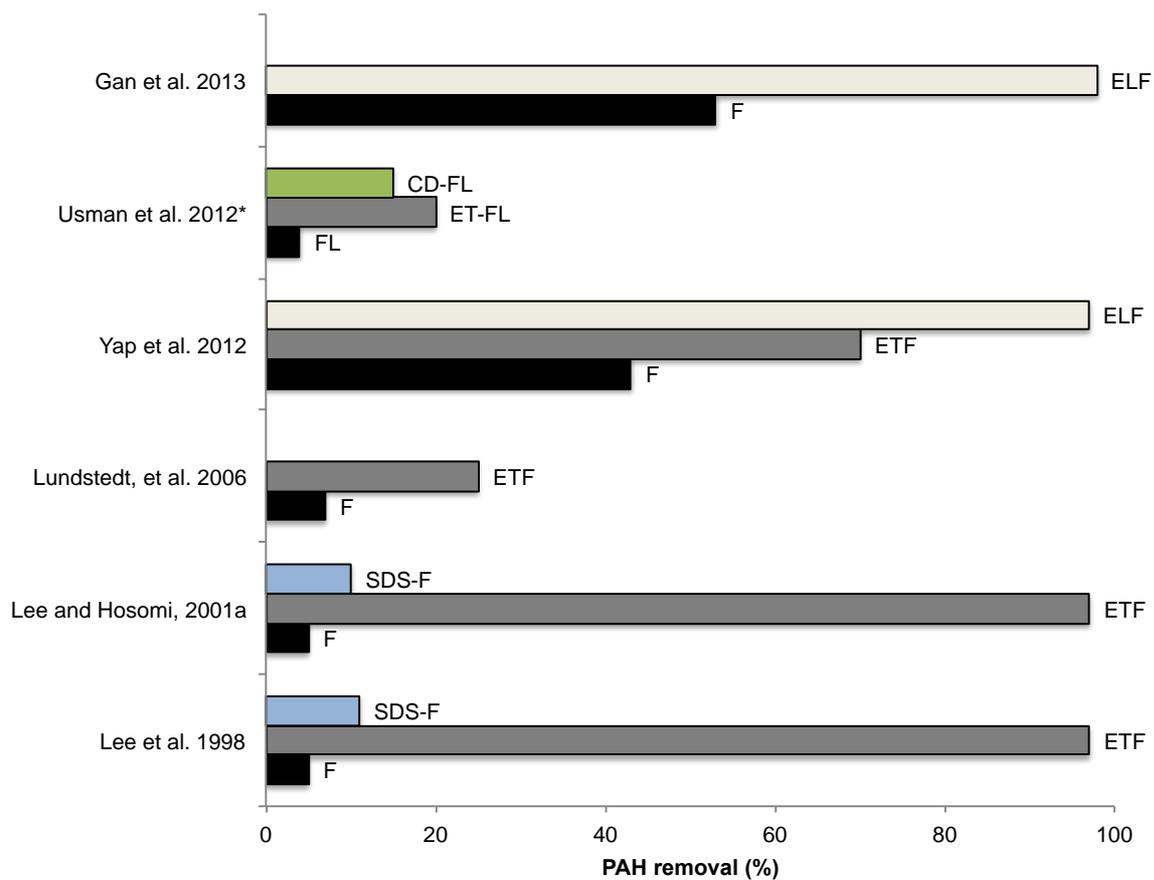
958

959 **Figure 3:**



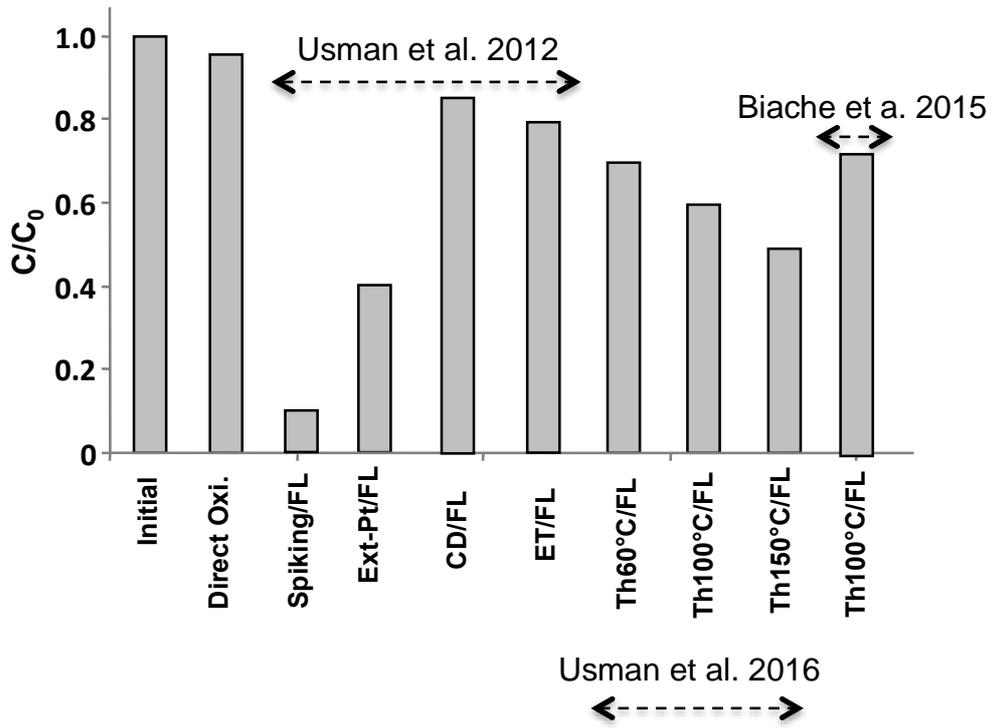
960

961 **Figure 4:**



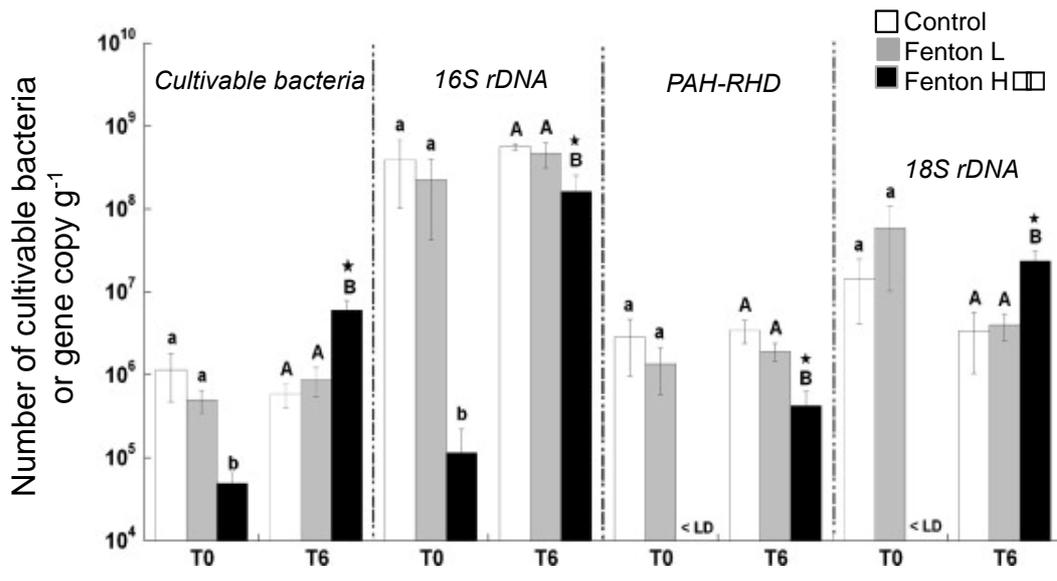
962

963 **Figure 5:**



964

965 **Figure 6:**



966