Integrative isotopic and molecular approach for the diagnosis and implementation of an efficient *in-situ* enhanced biological reductive dechlorination of chlorinated ethenes

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Abstract

Based on the previously observed intrinsic bioremediation potential of a site originally contaminated with perchloroethene (PCE), field-derived lactate-amended microcosms were performed to test different lactate isomers and concentrations, and find clearer isotopic and molecular parameters proving the feasibility of an *in-situ* enhanced reductive dechlorination (ERD) from PCE-to-ethene (ETH). According to these laboratory results, which confirmed the presence of *Dehalococcoides* sp. and the *vcrA* gene, an *in-situ* ERD pilot test consisting of a single injection of lactate in a monitoring well was performed and monitored for 190 days. The parameters used to follow the performance of the ERD comprised the analysis of i) hydrochemistry, including redox potential (Eh), and the concentrations of redox sensitive species, chlorinated ethenes (CEs), lactate, and acetate; ii) stable isotope composition of carbon of CEs, and sulphur and oxygen of sulphate; and iii) 16S rRNA gene sequencing from groundwater samples. Thus, it was proved that the injection of lactate promoted sulphate-reducing conditions, with the subsequent decrease in Eh, which allowed for the full reductive dechlorination of PCE to ETH in the injection well. The biodegradation of CEs was also confirmed by the enrichment in ^{13}C and carbon isotopic mass balances. The metagenomic results evidenced the shift in the composition of the microbial population towards the predominance of fermentative bacteria. Given the success of the *in-situ* pilot test, a full-scale ERD with lactate was then implemented at the site. After one year of treatment, PCE and trichloroethene were mostly depleted, whereas vinyl chloride (VC) and ETH were the predominant metabolites. Most importantly, the shift of the carbon isotopic mass balances towards more positive values confirmed the complete reductive dechlorination, including the VC-to-ETH reaction step. The combination of techniques used here provides complementary lines of evidence for the diagnosis of the intrinsic biodegradation potential of a polluted site, but also to monitor

48 the progress, identify potential difficulties, and evaluate the success of ERD at the field
49 scale.

50

51 **Keywords:** Chlorinated ethenes; Organohalide-respiring bacteria; *Dehalococcoides*; *In-*
52 *situ* pilot test; Enhanced reductive dechlorination; Carbon isotopic mass balance.

53

1. Introduction

Chlorinated ethenes (CEs) are among the most ubiquitous anthropogenic groundwater contaminants due to their widespread use in industry and recalcitrance under oxic conditions. CEs are considered priority substances (ATSDR, 2017) and have maximum contaminant levels in groundwater set by the 2008/105/EC European Directive (European Commission, 2008). Perchloroethene (PCE) is resistant to aerobic degradation but under anoxic conditions can undergo reductive dechlorination to the less-chlorinated ethenes trichloroethene (TCE), *cis*- and *trans*-dichloroethene (*cis*-, *trans*-DCE), vinyl chloride (VC), and the harmless end-product ethene (ETH).

Organohalide-respiring bacteria (OHRB) provide a potential solution to detoxify sites impacted with CEs due to their capability to use organochlorides as electron acceptors to support their growth, which result in a stepwise reduction of CEs (Bradley, 2000; Leys et al., 2013). Several OHRB can partially dechlorinate PCE to TCE or *cis*-DCE, e.g. *Geobacter* sp., but *Dehalococcoides* sp. (*Dhc*) and *Dehalogenimonas* sp. are the only genera, to date, capable of fully dechlorinating CEs to ETH (Adrian and Löffler, 2016; Yang et al., 2017b). Reductive dehalogenases (*rdh*) are the key enzymes driving organohalide respiration and can be used as biomarkers to investigate the intrinsic bioremediation potential at polluted sites (Blázquez-Pallí et al., 2019; Dugat-Bony et al., 2012; Hermon et al., 2019; Scheutz et al., 2008). Compound-specific stable isotope analysis (CSIA) can provide a complementary line of evidence to monitor biodegradation of chlorinated compounds in groundwater (Aelion et al., 2009; Hermon et al., 2018; Hunkeler et al., 1999; Nijenhuis et al., 2007; Palau et al., 2014). This technique measures a specific stable isotope ratio (e.g. $^{13}C/^{12}C$) within molecules and is based on light and heavy isotopes degrading at slightly different rates during biochemical transformations. Such shift in the isotopic composition (δ) of the molecule can be used to confirm and quantify *in-situ* biodegradation and distinguish degradation pathways (Elsner, 2010). This

is possible because the isotopic enrichment caused by physical processes such as volatilization, sorption, or dilution, is considered to be negligible (Aelion et al., 2009; Hunkeler et al., 1999). Similarly, the isotopic composition of certain non-halogenated potential electron acceptors can be used to trace changes in the redox potential of the groundwater. For instance, a dual isotope approach can be used to measure the extent of sulphate reduction because this reaction results in an enrichment in both ^{34}S and ^{18}O in the residual sulphate (Wu et al., 2011).

When OHRB are present but the electron donor becomes a limiting factor, the groundwater contaminated with CEs can show little to no dechlorination past *cis*-DCE, resulting in the accumulation of toxic intermediates (DCE or VC stall) (Bradley, 2000; Stroo and Ward, 2010). To avoid this, groundwater can be conditioned with organic fermentable substrates (e.g. lactate), which can generate reducing equivalents that promote the sequential dechlorination to ETH. This bioremediation approach is commonly referred to as biostimulation or ERD, which stands for enhanced reductive dechlorination (Adrian and Löffler, 2016; Leeson et al., 2004). To date, many studies have focused on laboratory methodologies to assess and characterize the intrinsic bioremediation potential of CEs-polluted sites by OHRB (Buchner et al., 2015; Courbet et al., 2011; Ebert et al., 2010; Kuder et al., 2013; Lee et al., 2016; Lu et al., 2009; Matteucci et al., 2015; Nijenhuis et al., 2007; Slater et al., 2001; Tarnawski et al., 2016; Yu et al., 2018), but few have reported results after applying ERD and CSIA at the field scale (Herrero et al., 2019; Hirschorn et al., 2007; Song et al., 2002).

The research reported in Blázquez-Pallí et al. (2019) presented a multi-method approach to assess the intrinsic biodegradation potential of an industrial site in Barcelona (Spain) polluted by CEs. Obtained results discouraged natural attenuation as a remediation strategy due to *cis*-DCE stall, and recommended an ERD injecting lactate as electron donor. In line with those conclusions, the present study aimed at continuing the

106 remediation work at that same aquifer and apply an integrated isotopic and molecular
107 approach that provides complementary lines of evidence to implement, monitor and
108 assess an efficient *in-situ* biodegradation of CEs at the site during an ERD. To this end, a
109 more detailed laboratory study focusing, as well, on the concentration and isomeric form
110 of lactate required for the complete dechlorination of PCE to ETH was performed first,
111 followed by the monitoring of the *in-situ* ERD pilot test and the final full-scale treatment.
112 The used methodology in all cases combined (1) the acquisition of hydrogeochemical
113 data, (2) microcosms experiments, (3) molecular techniques (i.e. identification of selected
114 biomarkers and 16S rRNA high-throughput sequencing of groundwater samples), and (4)
115 stable isotopes of sulphate (as model redox sensitive species), and carbon from the target
116 contaminants (CEs).

117

118 **2. Materials and methods**

119 *2.1. Chemicals*

120 PCE (99.9% purity) was from Panreac; laboratory-grade sodium DL-lactate (≥98%
121 purity) from Sigma-Aldrich (hereinafter, Lactate-1), and food-grade sodium L-lactate
122 (97% purity, at 60% w/w) from Purac (Corbion) (hereinafter, Lactate-2). Other chemicals
123 and reagents used for the present study were purchased from Sigma-Aldrich,
124 Thermofisher and Bio-Rad at scientific grade or higher.

125 *2.2. Study site*

126 The studied site is located in the Barcelona province (Spain) and the aquifer is mainly
127 constituted by three lithological units: i) a lower layer of reddish clay loams, ii) an
128 intermediate layer of brown and silty clays, and iii) an upper layer of ochre silty clays.
129 Given such lithology, the hydraulic conductivity is generally low, only improved by
130 localized gravel and sandstone areas that increase the permeability of the media and could
131 act as preferential flow paths. A hydrogeological cross section of the site is depicted in

Figure S1, and a more detailed description of the aquifer can be found elsewhere (Blázquez-Pallí et al., 2019). A preliminary site characterization revealed a significant PCE plume, accompanied by minor amounts of TCE, *cis*-DCE and VC. Originally, it was treated by a combination of pump and treat (P&T) and dual-phase extraction (DPE, vapour and groundwater). Pumped groundwater was later treated through an air stripping system. Groundwater flowed naturally in the NW–SE direction (Figure 1); however, under such ongoing remediation, all the extraction points were almost dried due to the low productivity of the aquifer. These P&T and DPE systems were halted partially during the *in-situ* pilot test (only remained active in some wells outside the pilot test area), and completely for the full-scale bioremediation. However, certain influence of this pumping on the groundwater flow direction during the *in-situ* pilot test cannot be discarded.

2.3. Microcosm experiments

Groundwater with fine sediments was collected from the intended pilot scale injection well PZ-2 (Figure 1) with a peristaltic pump on October 4th, 2016, in transparent sterile glass bottles that were sealed with PTFE caps as described elsewhere (Blázquez-Pallí et al., 2019). These groundwater samples were kept in the dark at 4ºC until the following day, when microcosms were set up. Four different treatments were prepared, at least in duplicate: (1) control containing only groundwater, (2) groundwater with Lactate-1 at ~ 3 mM, (3) groundwater with Lactate-2 at ~ 3 mM, and (4) groundwater with Lactate-2 at ~ 15 mM. Each 100 mL microcosm bottle contained 65 mL of sampled groundwater with fine sediments and the lactate concentrations described above. All microcosms were prepared in an anoxic glovebox and incubated in the dark at 25 °C. Microcosms that fully dechlorinated CEs were reamended with TCE and transferred to sterilized anoxic synthetic medium (3–7% v/v) described elsewhere (Martín-González et al., 2015) during the exponential degradation phase of CEs.

For the analysis of the site-specific carbon isotopic fractionation (εC) during degradation of PCE, six parallel anoxic cultures were prepared with the abovementioned defined medium and groundwater from PZ-2 (1.5% v/v) as inoculum. Each culture was spiked with PCE (235 µM), and sacrificed with NaOH (10 M) at 0, 1, 6, 77, 88, 96 and 97% of PCE degradation. Three different controls were prepared in duplicate: (1) killed controls with PCE, (2) killed controls without PCE, and (3) abiotic controls with PCE.

2.4. Implementation of the ERD in-situ pilot test

The pilot test consisted of a unique injection of lactate at well PZ-2 (Figure 1) on October 25th, 2016 (t_0). The product injected was an aqueous solution of Lactate-2 diluted with groundwater outflowing from the air stripping system installed at the site. The total volume of substrate injected was lower than 10% of the treatment zone volume. The design parameters of the *in-situ* pilot test were based on the recommendations from Leeson et al. (2004) and Dugat-Bony et al. (2012).

2.5. Monitoring of the ERD in-situ pilot test

Field parameters (i.e. Eh, pH, T, and electric conductivity (EC)) were measured *in-situ* and groundwater was collected with a peristaltic pump from the injection well PZ-2 and nearby wells PZ-1, PZ-3, PZ-6, PZ-22, MW-6, and Prof.D (Figure 1) as described elsewhere (Blázquez-Pallí et al., 2019). Sampling campaigns were carried out the day before injection of lactate (t_{-1}, October 24th, 2016) and the next 2, 9, 20, 50, 86, 142 and 190 days after the injection (hereinafter, t_i). Samples collected for both CSIA and CEs concentration were immediately killed with NaOH (pH>10) and stored at 4ºC until analysed. Short-chain volatile fatty acids (VFAs) were analysed from groundwater samples filtered on site (0.20 µm) and stored in borosilicate tubes at 4ºC until analysed.

2.6. Isotopic evaluation of the full-scale ERD treatment

A full-scale ERD treatment was implemented by Litoclean, S.L. on August 2017. The bioremediation strategy consisted in the injection of Lactate-2 every three months for the

period of a year. Between 30 and 50 out of a total of 66 monitoring wells at the site were used for the injection of the substrate at every event but changing the distribution of the injections to ensure the maximum coverage of the plume area. On September 2018, after one year of treatment, groundwater was sampled from wells PZ-3, PZ-5, PZ-22, MW-3, MW-6, and MW-7 (Figure 1) following the same methodology mentioned in section 2.5. Samples were analysed for CEs concentrations and CSIA, and data were compared to previous (before the *in-situ* pilot test) chemical and isotopic values for each well.

2.7. Analytical methods

CEs concentrations were analysed from 500 µL headspace samples by gas chromatography (GC) coupled to a flame ionization detector (FID) as reported by Martín-González et al. (2015). VFAs (lactate, pyruvate, acetate, formate) were analysed by high performance liquid chromatography (HPLC) from filtered liquid samples as described elsewhere (Mortan et al., 2017).

Stable carbon isotopes of CEs were analysed with an Agilent 6890 GC coupled to an IRMS at *Centres Científics i Tecnològics de la Universitat de Barcelona* (CCiT-UB), following the procedure described in Blázquez-Pallí et al. (2019). Carbon isotopic compositions in all samples are presented in delta notation (δ^{13}C, in ‰), relative to the international standard Vienna Pee Dee Belemnite (VPDB), following

$$\delta^{13}C = \left(\frac{R_{sample}}{R_{std}} - 1\right) \cdot 1000 \qquad (1)$$

where R_{sample} and R_{std} represent the isotope ratios (e.g. $^{13}C/^{12}C$) of the sample and the standard, respectively (Elsner, 2010). Instrument uncertainty was considered as the standard deviation (1σ) of duplicate measurements. For field data, the degradation is considered significant if the shift in δ^{13}C is >2‰ compared to its original value (Hunkeler et al., 2008).

Carbon isotopic mass balance for CEs at each monitoring well was calculated according to Aeppli et al. (2010) and Hunkeler et al. (1999), as follows

$$\delta^{13}C_{sum} = x_{PCE} \cdot \delta^{13}C_{PCE} + x_{TCE} \cdot \delta^{13}C_{TCE} + x_{DCE} \cdot \delta^{13}C_{DCE} + x_{VC} \cdot \delta^{13}C_{VC} \quad (2)$$

where *x* is the molar fraction of each substance with respect to the total molar mass (sum of CEs for which δ^{13}C values were available) for each sample at each sampling event. In this case, as PCE was the only contaminant spilled in this aquifer, the $\delta^{13}C_{sum}$ must remain constant as long as (1) PCE released along the time and space had the same isotopic composition, (2) the unique transformation pathway was reductive dechlorination, (3) VC does not further degrade to ETH when considered in the balance (Aeppli et al., 2010; Hunkeler et al., 1999; Palau et al., 2014).

The logarithmic form of the simplified Rayleigh equation correlates changes in the carbon isotope ratios (R_t/R_0) and changes in concentrations ($f=C_t/C_0$) with time for a closed system (Elsner, 2010), and the obtained epsilon (εC) represents the carbon isotopic fractionation, as follows

$$\ln\left(\frac{R_t}{R_0}\right) = \varepsilon C \cdot \ln(f) \quad (3)$$

where R_t/R_0 can be expressed as $(\delta^{13}C_t + 1) / (\delta^{13}C_0 + 1)$.

The analysis of major anions (HCO_3^-, NO_3^-, Cl^-, SO_4^{-2}) and cations (Na^+, K^+, Ca^{+2}, Mg^{+2}) was performed at CCiT-UB. Total concentrations of Na^+, K^+, Ca^{+2}, Mg^{+2} were analysed by inductively coupled plasma-optic emission spectrometry (ICP-OES, Optima 3200 RL) and by inductively coupled plasma mass spectrometry (ICP-MS, Elan 6000). NO_3^-, Cl^-, and SO_4^{-2} concentrations were determined by HPLC using a WATERS 515 HPLC pump with an IC-PAC anion column and a WATERS detector (mod 432), while HCO_3^- was measured by titration (METROHM 702SM Titrino). The predominant

equilibrium systems controlling the Eh were investigated via Eh–pH predominance diagrams prepared with the MEDUSA code (Puigdomènech, 2010).

Dissolved SO_4^{-2} detected in groundwater was precipitated as $BaSO_4$ as reported elsewhere (Dogramaci et al., 2001) and its sulphur and oxygen isotopic compositions were analysed following Rodríguez-Fernández et al. (2018). Results are presented in delta notation (δ^{34}S-SO_4^{-2} and δ^{18}O-SO_4^{-2}, in ‰), relative to the international standards, Vienna Standard Mean Oceanic Water (VSMOW) for δ^{18}O and Vienna Canyon Diablo Troillite (VCDT) for δ^{34}S.

2.8. DNA extraction, PCR and 16S rRNA gene high-throughput sequencing

DNA for molecular analyses was extracted from TCE-enriched field-derived cultures (three transfers into fresh synthetic medium) that were originally amended with Lactate-2 (~ 3 mM) and from well PZ-2. Cell harvesting, genomic DNA isolation and PCR reaction for amplification of the *vcrA* gene were performed as described in Blázquez-Pallí et al. (2019). Primer sets used are detailed in Table S1.

To investigate the OHRB involved in the biodegradation of CEs to ETH, the dilution-to-extinction method (Löffler et al., 2005) was applied in 20-mL vials containing 12 mL of the anoxic synthetic medium mentioned in section 2.3, but using 1 mL of active TCE-enriched field-derived culture as inoculum and *cis*-DCE as electron acceptor. After three extinction series, the more diluted vial showing ETH formation was used as inoculum for serum bottle microcosms and, after consuming 10 µM *cis*-DCE, was selected for 16S rRNA analysis. After DNA extraction, amplicons of the region V3–V4 for 16S rRNA genes were amplified with primers S-D-Bact-0341-b-S-17 and S-D-Bact-0785-a-A-21 (Klindworth et al., 2013) with the Illumina MiSeq sequencing platform at *Serveis de Genòmica i Bioinformàtica* from *Universitat Autònoma de Barcelona* (Spain).

To characterize the impact of the injection of lactate on the bacterial community

structure after the *in-situ* pilot test, 80 mL of groundwater were collected from wells PZ-2, PZ-1, and PZ-22 at t_{-1} and t_{142}. Samples were stored at -20ºC until DNA extraction and 16S rRNA gene high-throughput sequencing were performed at AllGenetics & Biology (A Coruña, Spain). For DNA extraction, samples were centrifuged at 5000 *g* for 1 h and the pellet was transferred to PowerBead tubes of the DNeasy Powersoil DNA isolation kit (Qiagen). DNA was isolated following the instructions of the manufacturer. For library preparation, a fragment of the bacterial 16S rRNA region of around 450 bp was amplified using primers Bakt_341F and Bakt_805R (Herlemann et al., 2011). The pool was sequenced in a MiSeq PE300 run (Illumina).

3. Results and discussion

3.1. Feasibility and markers for complete ERD at laboratory scale

3.1.1. Assessment of the lactate isomers and concentration in microcosm experiments

Microcosms were prepared to test whether reductive dechlorination of CEs to ETH was feasible in the injection well PZ-2. Besides the control, which accounted for monitored natural attenuation (MNA), two different isomeric forms of sodium lactate (DL-/Lactate-1 and L-/Lactate-2) were used to test their effect on the lactate fermentation potential of the native microbial community. Moreover, Lactate-2 was used because it was a candidate product in a foreseeable *in-situ* pilot test. Accordingly, Lactate-2 was amended at two different concentrations to discard inhibition effects. On the one hand, the unamended controls transformed PCE to *cis*-DCE and VC within 15 days, but ETH was not detected. When microcosms were reamended with TCE (35 µM) at day 25, dechlorination barely passed VC, and ETH was detected at low concentration after 50 days (Figure S2A). On the other hand, lactate-amended treatments fully dechlorinated PCE to ETH within 15 days and exhibited similar rates independently of the isomeric form and concentration (Figure S2, B–D). After adding TCE in all three of them at days

281 25 and 45, the dechlorination remained active with the consequent accumulation of ETH
282 (Figure S2, B–D). In all amended microcosms, lactate was totally consumed and acetate
283 was produced within 10 days (data not shown). However, excess of electron donor and,
284 thus, of H_2, did promote methanogenesis, resulting in a vigorous formation of methane in
285 the microcosm with ~ 15 mM of lactate (Figure S3), which has been reported before
286 (Blázquez-Pallí et al., 2019; Leeson et al., 2004).

287 3.1.2. Identification of *vcrA* gene and CEs dechlorinating bacteria

288 PCR amplifications with the *vcrA* gene-targeted primer were run to investigate
289 whether the *vcrA* gene was present in the TCE-enriched field-derived cultures from well
290 PZ-2 (section 2.8). After gel electrophoresis, observed diagnostic amplicons indicated
291 that the culture contained *vcrA* gene, which is implicated in the VC-to-ETH
292 dechlorination step (Figure S4).

293 In these TCE-enriched cultures, three dilution-to-extinction series were applied using
294 *cis*-DCE as electron acceptor to get insight into the OHRB responsible for ETH
295 generation. The 16S rRNA gene amplicon sequencing and the taxonomic assignments
296 revealed that the four predominant genera were *Pleomorphomonas* (50%), described as
297 nitrogen-fixing bacteria (Im et al., 2006; Madhaiyan et al., 2013; Xie and Yokota, 2005);
298 *Desulfomicrobium* (18%), which are sulphate-reducing bacteria (Sharak Genthner et al.,
299 1997, 1994); and the OHRB *Geobacter* (14%) and *Dehalococcoides* (3%) (Figure 2).
300 *Geobacter* sp. can derive energy from acetate oxidation coupled to PCE-to-*cis*-DCE
301 dechlorination, but it is also capable of growing on a wide range of non-halogenated
302 electron acceptors (Atashgahi et al., 2016; Sanford et al., 2016). The presence of
303 *Dehalococcoides* was consistent with the detection of *vcrA* reductive dehalogenase gene
304 (Figure S4) and ETH generation in lactate-amended microcosms (Figure S2).

305

306 3.1.3. Site-specific εC for PCE degradation

307 The analysis of site-specific εC can help estimate the extent of the *in-situ*
308 biodegradation of contaminants in the field (Elsner, 2010). PCE was depleted in the
309 microcosms inoculated with groundwater from PZ-2 (Figure S5A) but dechlorination was
310 not accompanied by a significant change in its isotopic composition ($\delta^{13}C_{PCE}$) (Figure
311 S5B). In more detail, $\delta^{13}C_{PCE}$ shifted only 0.81‰ and 1σ for duplicate measurements
312 were, for all samples, below total instrumental uncertainty of 0.5‰ (Sherwood Lollar et
313 al., 2007). The dechlorination reaction did not fit the Rayleigh model (Eq. 3, $R^2 = 0.63$,
314 $\varepsilon C = -0.2 \pm 0.2‰$), which deemed this site-specific εC as not significant (ns). In contrast,
315 a stronger enrichment in ^{13}C was observed when the produced TCE was further degraded
316 to *cis*-DCE (Figure S5B). This non-linear εC for PCE is, most likely, the combined effect
317 of several OHRB simultaneously transforming PCE, with a major contribution of the non-
318 fractionating species. This agrees with the predominance of *Geobacter* sp. in the cultures
319 (Figure 2), which is reported to have a non-significant εC for PCE and could have lead
320 the PCE-to-TCE dechlorination (Table S2) (Cichocka et al., 2008). Abiotic and NaOH-
321 killed controls did not show PCE losses or degradation as its concentration did not vary
322 significantly throughout the whole experiment (236 ± 8 μM, n=6, Figure S5A) and no
323 degradation products were detected.

324 *3.2. In-situ ERD pilot test with lactate at the injection well PZ-2*

325 Microcosms of PZ-2, which mimicked the natural attenuation conditions at the site,
326 indicated that reductive dechlorination could stall at an intermediate stage, resulting in an
327 accumulation of *cis*-DCE and VC (Figure S2A). However, the amendment of lactate
328 resulted in a faster dechlorination of PCE and concomitant generation of ETH (Figure S2,
329 B–D). Considering these results, an *in-situ* ERD pilot test with Lactate-2 was
330 implemented at well PZ-2.

3.2.1. Hydrochemistry changes induced by the lactate injection

The concentration of major anions (SO_4^{-2}, NO_3^-, HCO_3^-, Cl^-) and cations (Na^+, K^+, Ca^{+2}, Mg^{+2}) was analysed at t_{-1}, t_{20}, t_{86} and t_{190}. At the injection well PZ-2, NO_3^- and SO_4^{-2} were depleted after the addition of the Lactate-2 solution (Figure S6A), whereas Na^+, Ca^{+2}, Mg^{+2} and electric conductivity (EC) increased, responding to the injected electron donor (Table S3). At the monitoring wells PZ-1, PZ-6 and MW-6, NO_3^- concentrations were depleted as well, but different trends were observed for SO_4^{-2} (Figure S6, A and B). In contrast, no significant changes were observed for those anions in monitoring wells PZ-3, PZ-22, and Prof.D (Figure S6C).

Geochemical modelling shows that the aquifer was at nitrate-reducing conditions before injection and that the addition of lactate promoted a shift towards sulphate-reducing and methanogenic conditions (Figure S7). This shift was more extreme in the injection well PZ-2 (at t_{20}) as well as in monitoring wells PZ-1 (at t_{86}), and PZ-6 and MW-6 (both at t_{190}), which agrees with the decrease observed in NO_3^- and SO_4^{-2} concentrations (Figure S6). In addition, results suggested that the system was controlled by calcite ($CaCO_3$(s)) (Figure S7B), as the pH was maintained within the range of 6–8, the optimal for OHRB (Yang et al., 2017a).

$\delta^{34}S$ and $\delta^{18}O$ values of dissolved SO_4^{-2} from t_{-1} at each monitoring well were compared with the ones obtained after the lactate injection (t_{20}, t_{86} or t_{190}). Before the injection, the S and O isotopic composition of SO_4^{-2} at the site showed diverse values corresponding to a mixture among geogenic composition (Triassic or Tertiary recycled gypsum) in well Prof.D, synthetic fertilizers, and an unknown source (with the lowest ^{34}S and ^{18}O values) that could be related to agricultural uses of manure (Otero et al., 2008). This variability points out to a heterogeneity in the origin of the dissolved sulphate in the aquifer (Figure 3). In any case, in the injection well PZ-2, isotopic compositions at t_{20} became significantly more enriched with a shift of $\Delta\delta^{34}S$ = +3.0‰ and $\Delta\delta^{18}O$ = +3.2‰

resulting in a slope of 1.1 (Figure 3). These $\delta^{34}S$ and $\delta^{18}O$ values of sulphate seem influenced by the mixing with the sulphate isotopic composition of the injected water (Figure 3). Unluckily, no values could be obtained for t_{86} or t_{190} due to concentrations being below the limit of quantification. An even larger shift was measured in monitoring wells PZ-6 and MW-6 because the slower decrease of SO_4^{-2} concentrations allowed the measurement of more points. By t_{190}, PZ-6 showed the most ^{34}S enriched value measured at the site, also with respect to its t_{-1} ($\Delta\delta^{34}S$ = +14.4‰), but with a small $\delta^{18}O$ enrichment ($\Delta\delta^{18}O$ = +1.0‰) (Figure 3). For MW-6, the most enriched $\delta^{34}S$ was measured in t_{86} ($\Delta\delta^{34}S$ = +7.4‰), while the largest $\Delta\delta^{18}O$, of +4.7‰, was measured in t_{190} (Figure 3). In contrast, $\Delta\delta^{34}S$ and $\Delta\delta^{18}O$ in monitoring wells PZ-1, PZ-3, PZ-22, and Prof.D were not significant (Figure 3). A wide variation in the dual S–O slopes have been described for bacterial sulphate reduction in different natural environments (from 0.23 to 1.7) (Antler et al., 2013; Mizutani and Rafter, 1973). These differences have been linked to the net sulphate reduction rate and to the recycling of intermediate species (such as sulphite that facilitate oxygen isotope exchange with H_2O) back to sulphate. In environments where sulphate reduction is fast, this sulphite re-oxidation is minimal resulting in lower slopes due to sulphur isotopes increasing faster than oxygen isotopes (Antler et al., 2013). Thus, obtained results (with the available data points) proved sulphate reduction after lactate injection only in PZ-6 and MW-6 by isotopic enrichment rather than pure mixing with injected water.

3.2.2. Enhanced biodegradation of chlorinated ethenes at the injection well PZ-2

PCE was the main toxic substance dissolved in groundwater of PZ-2 before lactate injection (Figure 4A). After biostimulation, Eh decreased from +300-400 mV to ~ -50 mV by t_{20} (Figure 4D) and acetate was detected for the first time by t_{50} (Figure 4C). The generation of reducing equivalents from lactate fermentation and the subsequent dramatic decrease of Eh favoured dechlorination past *cis*-DCE and VC up to ETH by t_{190} (Figure

383 4A). The full dechlorination of CEs to ETH by t_{190} was also confirmed by the enrichment
384 in the isotopic mass balance ($\delta^{13}C_{sum}$), which changed from -33 ± 2‰ to -9.0 ± 0.5‰
385 (Figure 4B). This difference in the $\delta^{13}C_{sum}$ is due to not considering the isotopic
386 composition of ETH after VC degradation (which would be depleted as lighter isotopes
387 react faster). Hence, $\delta^{13}C_{sum}$ became less negative as VC to ETH reaction progressed. In
388 detail, the PCE to TCE reaction at PZ-2 did not change $\delta^{13}C_{PCE}$ significantly and remained
389 constant throughout the whole monitoring period (Figure 4B), emulating the results of
390 the microcosm experiments (εC = ns) (section 3.1.3.). Conversely, TCE and *cis*-DCE
391 showed variations in $\delta^{13}C$ during the *in-situ* dechlorination process, which were up to
392 +7‰ for TCE and +16‰ for *cis*-DCE at the measured times (Figure 4B). Lastly, VC
393 could be nicely traced, exhibiting a remarkable variation of $\delta^{13}C$ (shift up to +49‰)
394 including its formation from *cis*-DCE and degradation to ETH by t_{190} (Figure 4B).

395 3.2.3. Impact of the *in-situ* ERD pilot test at wells within the direct radius of influence

396 Lactate and/or acetate were progressively detected in PZ-1 (at t_2), PZ-6 (at t_{20}) and
397 finally in MW-6 (at t_{142}), indicating the arrival of the injected solution through the
398 preferential groundwater flow paths (with a stronger N–S direction than at natural
399 conditions, Figure 1), thus drawing the direct radius of influence of the *in-situ* ERD pilot
400 test. Another evidence of the affectation at these wells was the dramatic decrease in Eh
401 like the one observed in the injection well PZ-2 (Figures S8, S9, S10). However, even
402 when reaching similar negative Eh values (down to -100 mV), the PCE reductive
403 dechlorination mainly stalled at *cis*-DCE (most evident in PZ-1) (Figures 5, S8, S9, S10),
404 while in PZ-6 (farther from PZ-2) *cis*-DCE started to pass to VC and ETH by t_{190} (Figure
405 S9A). These concentration data agree with $\delta^{13}C_{sum}$ of CEs remaining constant in the three
406 wells (~ -32‰ over 190 days, Figures 5, S8B, S9B, S10B), so there was no significant
407 alternative degradation pathways besides ERD, and ETH generation was still minor
408 compared to PZ-2. Curiously, the most inefficient ERD (*cis*-DCE stall confirmed by no

changes in its $\delta^{13}C_{cis\text{-DCE}}$ after t_{50}) of these wells occurred in PZ-1, where the initial drop in the Eh was followed by a rebound to more positive values after t_{86} (Figure S8D). In fact, as said before, according to sulphate isotopic composition, PZ-1 did not reach significant sulphate reducing conditions (Figure 3). This was most likely caused by the consumption and absence of electron donor from t_{86} onwards or fast entrance of oxidizing groundwater. In contrast, possibly due to the combination of preferential flow paths and the influence of external active remediation systems (P&T and DPE) on the groundwater flow direction, certain amounts of organic acids arrived to wells PZ-6 and MW-6 (Figure S9C, S10C), which were the farthest from PZ-2. Their Eh decreased much more progressively after t_{50}, and remained negative until t_{190}, allowing sulphate reducing conditions and further degradation to VC (Figure S9D, S10D).

3.2.4. Impact of the *in-situ* pilot test in wells outside the direct radius of influence

Lactate and acetate were not detected in wells PZ-3, PZ-22, and Prof.D (Figures S11C, S12C and S13C). Accordingly, Eh did not reach negative values in any of them (Figures S11D, S12D and S13D), not allowing to prove a direct effect of the ERD pilot test. In well PZ-3, Eh decreased to +60 mV at t_{86} but it rebounded to initial conditions at t_{190} (Figure S11D). Curiously, CEs concentrations decreased globally in PZ-3 and PZ-22, but PCE molar fraction increased in respect to its daughter products, which could indicate a solubilization and dilution of the PCE adsorbed to the soil due to the injected volume (Figure S14). Therefore, the ^{13}C enrichments observed for TCE and *cis*-DCE in PZ-3 (by +11.5‰ and +8.1‰, respectively, Figure S11) could be attributed to reductive dechlorination in the well or, most probably, associated to the dispersion from upstream contaminants (Figure 1). In wells PZ-22 and Prof.D, changes in CEs concentrations (Figures S12A and S13A) were not associated to significant shifts in ^{13}C values (Figures S12B and S13B), confirming this dilution effect.

435 3.2.5. Changes in the native microbial community induced by the *in-situ* pilot test

436 The effect of lactate injection on the microbial community of the aquifer was
437 investigated by high-throughput sequencing of the 16S rRNA region of selected
438 groundwater samples. PZ-2, PZ-1 and PZ-22 at t_{-1} and t_{142} were chosen based on the
439 results obtained in the pilot test, as they represented three different scenarios: i) PCE-to-
440 ETH reaction at the injection well (PZ-2), ii) *cis*-DCE stall (PZ-1), and iii) a well not
441 directly impacted by the injection of lactate (PZ-22). In PZ-2, the most abundant phyla at
442 t_{-1} were *Proteobacteria* (58%), *Planctomycetes* (11%), *Bacteroidetes* (6%),
443 *Verrucomicrobia* (6%), *Chlamydiae* (4%), *Actinobacteria* (4%), and *Acidobacteria* (2%),
444 whereas at t_{142}, the community shifted to *Firmicutes* (67%), *Bacteroidetes* (14%),
445 *Proteobacteria* (14%), and *Tenericutes* (2%) (Figure 6, Table S4). Within *Firmicutes*,
446 bacteria from the family *Veillonellaceae* (26%), and *Erysipelotrichaceae* (9%) were the
447 most abundant; and the same occurred for the genus *Desulfovibrio* (3%), from the
448 *Proteobacteria* phylum (data not shown). The microbial composition after the injection
449 of lactate was dominated by fermenting bacteria that were probably induced by the
450 injection of lactate (Fennell et al., 1997; He et al., 2007; Tegtmeier et al., 2016). In
451 contrast, the abundance of the *Firmicutes* phylum was very limited in well PZ-1
452 (abundance of 4% at t_{142}) and almost insignificant in well PZ-22, which agrees with the
453 chemical and isotopic results discussed before.

454 *3.3. Isotopic evaluation of the full-scale ERD with lactate*

455 After the *in-situ* ERD pilot test, which finished in June 2017, a full-scale ERD with
456 Lactate-2 was implemented on August 2017. After one year of full-scale treatment (from
457 August 2017 to September 2018), an isotopic mass balance of CEs was calculated from
458 6 selected monitoring wells (PZ-3, PZ-5, PZ-22, MW-3, MW-6, MW-7, Figure 1) to
459 evaluate the extent and success of the full-scale ERD. Results revealed that PCE and TCE
460 were completely depleted in 5 out of the 6 monitoring wells and VC and ETH were the

major end products (Table 1). Originally, $\delta^{13}C_{sum}$ at the site was, in average, of -30 ± 2‰, which responded to the original $\delta^{13}C$ value of PCE (Blázquez-Pallí et al., 2019). After one year of full-scale ERD, the $\delta^{13}C_{sum}$ values in those 5 wells were enriched (ranging from -23.2 ± 0.5‰ to -13 ± 1‰) with respect to the initial value (Table 1). This trend in $\delta^{13}C_{sum}$ of becoming more positive was also observed in well PZ-2 during the *in-situ* pilot test and responds to ETH production. In the case of MW-7, the $\delta^{13}C_{sum}$ was already different from the rest of the wells in the characterization campaign (May 2016) (Blázquez-Pallí et al., 2019), but biodegradation up to VC has been also observed after the full-scale (Table 1). Therefore, these results confirmed that the reductive dechlorination of CEs was occurring across the site, albeit at different extent in each well.

4. Conclusions

The present study aimed at validating a multidisciplinary methodology to assess and monitor biodegradation of CEs at the field scale. In this work, hydrochemical, isotopic and microbiological data from the field was examined together with aquifer-derived laboratory microcosms to get an insight into the idiosyncrasies of the aquifer. The first part of this study focused on the diagnosis of the *in-situ* biodegradation potential of the site for full reductive dechlorination of CEs. The results obtained provided evidence that the stall at *cis*-DCE and VC was due to the lack of electron donors. Afterwards, the injection of lactate in a monitoring well evidenced that full reductive dechlorination to ETH was achieved within 190 days, and that the reaction followed a different pace in the surrounding monitoring wells. Therefore, the hydrodynamic characteristics of the aquifer possibly dominated the distribution of electron donor, hence, affecting the outcome of the biodegradation reaction in the studied surrounding wells. The methodology proposed here can be easily integrated as a tool in the early stages of site investigation to decide which bioremediation strategy is more suitable according to the specific characteristics of the

aquifer. Moreover, it can reduce the uncertainties regarding the ongoing processes that are occurring in groundwater systems. Finally, the initial isotopic characterization of the site allowed for applying isotope-mass balance calculations to unequivocally demonstrate ETH generation after the full-scale enhanced reductive dechlorination via lactate injection. This stable isotope-based technique can overcome the difficulties related to the traditional mass balance approaches which are often hindered by natural processes (e.g. dilution, sorption, volatilization).

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