The effect of common groundwater anions on the aqueous corrosion of zero-valent iron nanoparticles and associated removal of aqueous copper and zinc

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1. Introduction

Aqueous contaminant transformation and/or removal by nanoscale zero-valent iron particles (nZVI) is dependent on the aqueous corrosion pathway and fate of the particles. The corrosion process is dependent upon a range of different factors including time, dissolved oxygen (DO) concentration, organic matter concentration, pH and the concentration of common groundwater cations and anions [1,2].

To date, the majority of empirical studies have investigated contaminant removal onto nZVI using “chemically simple” solutions, i.e. deionised water plus the contaminant of interest, which is far removed from the geochemical complexity of natural waters. The influence of common groundwater anions on the removal and retention of aqueous contaminants by nZVI is therefore of great interest. The major groundwater anions are chloride (Cl−), bicarbonate (HCO3−) and sulphate (SO4 2−), and are typically present at concentrations within the range of 1.0−1000 mg L−1. Nitrate (NO3−) is less common and typically present at concentrations between 0.01−10 mg L−1. NO3− contamination is a growing concern in the UK and many other countries, however, and as such concentrations are recorded to be rising at an average of 0.34 mg NO3− L per year [3].

Both Cl−[2,4,5] and SO4 2−[2,5] increase iron corrosion rates by destabilising any passivating films which may form. In contrast NO3− inhibits corrosion [2,5–7], and also acts to nullify the corrosion-promoting effect of Cl−[8]. However, there have been limited studies investigating how such anions influence the corrosion pathway of nZVI. Reinsch et al. [9] exposed hydrogen-reduced nZVI to 10 mM solutions of Cl−, HCO3−, NO3− and SO4 2−. The passivation of nZVI was recorded in increasing order as HCO3−, SO4 2−, Cl− and NO3−. NO3− passivation was reported to be very influential: Fe0 was still detected after 6 months aqueous exposure. Moreover Kanel et al. [10] recorded HCO3−, NO3− and SO4 2− as not effecting As5+ uptake until anionic concentrations reached 10 mM. Furthermore Lim and Zhu [11] recorded NO3− as able to limit the...
breakdown of TCE by bimetallic Fe-Pd nanoparticles at concentrations as low as 2 mM, whereas HCO₃⁻ exerted only a minor inhibitory influence. SO₄²⁻ did not exhibit a significant influence.

The corrosion experiments outlined above were performed using hydrogen reduced nZVI, which has been shown to be markedly less reactive than borohydride reduced particles [12]. Moreover, the experimental time periods of these studies may not be of sufficient duration to ascertain final corrosion rates of nZVI when exposed to common groundwater anions. In this work, we aimed to bridge these gaps in our understanding by investigating the removal efficacy of Cu and Zn ions (two very common groundwater contaminants) onto borohydride reduced nZVI in solutions also containing major groundwater anions (namely: HCO₃⁻, Cl⁻, SO₄²⁻ and NO₃⁻).

2. Materials and methods

2.1. Chemicals

All chemicals (iron sulphate [FeSO₄·7H₂O], nitric acid (HNO₃), sodium borohydride (NaBH₄), sodium hydroxide (NaOH), sodium bicarbonate (NaHCO₃), sodium chloride (NaCl), sodium nitrate (NaNO₃), sodium sulphate (Na₂SO₄), copper acetate (Cu(CH₃COO)₂), zinc acetate (Zn(O₂CCH₃)₂) and solvents (ethanol, acetone) used in this study were of analytical grade.

2.2. nZVI synthesis

Pure nZVI was synthesised via the borohydride-reduction of ferrous iron [13]. First 7.6 g of Fe₂(SO₄)₃·6H₂O was dissolved in 200 mL of Milli-Q water (resistivity > 18.2 MΩ) that had been purged with oxygen free nitrogen gas for 10 min. The solution pH was then slowly adjusted to 6.8 by adding 4 M NaOH dropwise to prevent the formation of hydroxocarbonyl complexes. To this, 3.0 g of NaBH₄ was added (~0.2 g per min). The resultant solid material was cleaned through centrifugation and sequential rinsing in 50 mL of Milli-Q water, acetone and then ethanol. The nanoparticles were dried under low vacuum in a vacuum desiccator (~10⁻² mbar), and then stored in an argon filled (BOC, 99.998%) MBraun glovebox until required.

2.3. Experimental procedure

To assess changes to nZVI structure, Schott-Duran jars (2 L) were filled with 1.6 L of a 10 mM sodium-anion solution. The bicarbonate solution was synthesised from sodium bicarbonate (NaHCO₃), the chloride solution from sodium chloride (NaCl), the nitrate solution from sodium nitrate (NaNO₃) and the sulphate solution from sodium sulphate (Na₂SO₄). From each batch 100 mL of solution was removed, to which 1.6 g of nZVI was added. This was sonicated for 5 min in a VWR USC100T ultrasonic cleaner to disperse the nanomaterial, and then returned to the parent jar.

From these, 200 mL aliquots of nZVI-liquid suspensions were removed after 2 h, 1 d, 3 d, 1 w, 2 w, 4 w, 8 w, and 16 w.

Batch tests were performed with 400 mL solutions of Cl⁻, NO₃⁻ and SO₄²⁻ anions at 0.1, 1.0 and 10.0 mM, with 0.3 mM (~20 mg L⁻¹) of a metal solution and 0.5 g L⁻¹ nZVI. The influence of HCO₃⁻ on Cu and Zn uptake onto nZVI was not investigated due to the formation of insoluble Cu or Zn carbonate precipitates. The experimental matrix and starting pH and ORP (oxidation reduction potential) values are shown in Table 1. All batch tests were performed using Pyrex beakers. All batches were sonicated for 5 min after nZVI addition, and then covered loosely with Saragold plastic film to allow gas exchange but to prevent evaporation/cross contamination. From these, 10 mL aliquots of nZVI suspensions were taken at 0 h, 1 h, 2 h, 1 d, 3 d, 1 w, 2 w and 4 w. Geochemical parameters (pH, ORP and DO) were measured concurrently using a Hach Multimeter (model HQ40d).

To ensure a homogenous sample was taken, all sample containers were gently agitated prior to sampling. The extracted nZVI suspensions were separated into solid and liquid phases via 10 min of centrifugation at 6500 rpm using an Eppendorf 5810 centrifuge (nZVI corrosion experiments) or for one minute using a Hamilton Bell Vanguard V6500 desktop centrifuge (Cu and Zn uptake experiments). The liquid supernatant was then syringe filtered through a 0.22 μm cellulose filter, and then acidified using 13% v/v nitric acid to prevent metal precipitation or adsorption to the sample vessel. The solid material was washed by the same cleaning method described in Section 2.2, and then stored under ultra-pure ethanol until required for further analysis (~24 h).

2.4. Sample analysis methods

A Phillips X Pert Pro diffractometer with a Cu Kα radiation source (λ = 1.5406 Å) was used for X-ray diffraction (XRD) analysis (generator voltage of 40 keV; tube current of 30 mA). XRD plots were acquired between 10 and 70 °2θ with a step size of 0.02 °2θ and a 2 s dwell time. Plots were compared to the following Joint Committee on Powder Diffraction Standards (JCPDS) cards: goethite 81-0463, lepidocrocite 74-1877, maghemite 39-1346 and magnetite 86-1362. Miller indices for the identified peaks are shown in Table S1 (Supplementary material). A Kratos AXIS ULTRA-DLD spectrometer employing a monochromatic X-ray source (AlKα, 1486.6 eV) was used for X-ray photoelectron spectroscopy (XPS) analysis. Survey scans were acquired using a pass energy of 160 eV and a 1 eV step, whilst high resolution spectra were acquired using a 40 eV pass energy and a 0.1 eV step. Data analysis was performed using CasaXPS software after subtraction of a Shirley background, and fitted following the method of Grosvenor et al. [14]. All spectra were calibrated to the C1s line taken to be 285 eV for adventitious carbon. In preparation for BET specific surface area (SSA) analysis, samples were degassed under vacuum (~10⁻² mbar) for a 24 h period at a temperature of 75 °C. A known mass of this materials was then measured using a
Quantachrome NOVA 1200 surface area analyser, with N₂ as the adsorbent and following a 7 point BET method. Transmission electron microscopy (TEM) images of nZVI mounted on 200 mesh holey carbon coated Cu grids were obtained with a JEOL JEM 1200 EX Mk 2 TEM operating at 120 keV. Inductively coupled optical emission spectrometry (ICP-OES) was used to determine the aqueous concentrations of Cu, Zn and iron (Fe). The instrument used was an Agilent 710 ICP-OES. Blanks and standards were prepared in 1% nitric acid.

3. Results

3.1. Preliminary characterisation of the nZVI

Characterisation of the as-formed nZVI revealed features that are commonly associated with borohydride-reduced nZVI [15]. Analysis by XPS and XRD indicated that the core of nZVI was poorly crystalline α-Fe (characteristic peak at 44.9° 2θ) with a shell of mixed valent iron oxides/oxyhydroxides (Fig. 1). TEM analysis (Fig. 2) determined that the particles were roughly spherical, with a typical size range of 40–100 nm and no clear grain structure identified. BET analysis determined that the surface area of nZVI was 11.5 m² g⁻¹.

3.2. X-ray diffraction results from nZVI corrosion experiments

X-ray diffraction (XRD) plots for nZVI removed from the experimental solutions after 16 w are presented in Fig. 3. Previously published XRD data of nZVI removed from a solution without anions, but performed with the same experimental methodology, is also presented (hereafter, termed as the control) [16]. Time resolved XRD plots are presented in Figs. S1–S3 (Supplementary material). Crystal structure changes, as well as the relevant equations have been previously documented in Cornell and Schwertmann [17]. Crystal structural information for the identified minerals are shown in Table S2 (Supplementary material).

3.2.1. Influence of Cl⁻ on nZVI corrosion

The 10 mM Cl⁻ solution (Fig. S1, Supplementary material) was recorded to corrode the nZVI more rapidly than the control solution (solution of deionised water only), with the emergence of magnetite/maghemite (Fe₃O₄/γ-Fe₂O₃) after only 2 h (Fig. S3, Supplementary material). Due to the similar spinel crystal structure of Fe₂O₃ and γ-Fe₂O₃, their differentiation by XRD was not possible [9,18]. Lepidocrocite (γ-FeOOH), was identified after 1 w, and after 2 w, a small akaganéite (β-FeOOH) peak was observed. The latter iron requires chloride to form (alternative formula Fe₃⁺O(OH,Cl)) and proved to be transient, not appearing in any other scans. Zero-valent iron (44.9° 2θ) was detected for up to 2 w, compared to 4 w for the control system. Only minor changes were observed between the 4 w material and the final 16 w material (Fe₂O₃/γ-Fe₂O₃ and γ-FeOOH were the main phases), although a single α-FeOOH peak at 21.3° 20 was also identified (Fig. 3).

3.2.2. Influence of HCO₃⁻ on nZVI corrosion

After an initial emergence of Fe₂O₃/γ-Fe₂O₃ after 2 h (Fig. S2, Supplementary material), the predominant phase identified was goethite (α-FeOOH), which was detected as a minor contribution after 1 d in conjunction with Fe₂O₃/γ-Fe₂O₃. No iron carbonate

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Fig. 1. Fitted X-ray photoelectron plot with fitted photoelectron peaks (left) and X-ray diffraction plot with identified peaks for the range 10°–70° 2θ (right) acquired for the as-formed nZVI.

Fig. 2. TEM image acquired for the as-formed nZVI.

Fig. 3. X-ray diffraction plots with identified peaks for the range 10°–70° 2θ acquired for nZVI extracted from experimental solutions containing 10 mM Cl⁻, HCO₃⁻, NO₃⁻, SO₄²⁻ and control (anion-free) after 16 w (ascending). Key: G – α-FeOOH, L – γ-FeOOH and M – Fe₃O₄/γ-Fe₂O₃.
phases (e.g. siderite) were identified. The Fe⁰ peak (44.9° 2θ) was not detected after 3 d. Similar to the aforementioned Cl⁻ bearing batch system, little difference was recorded between the 4 and 8 w XRD profiles (Fig. 3), with α-FeOOH the main phase identified, along with Fe₃O₄γ-Fe₂O₃.

3.3.1. Reverted 3.3.1. Cl⁻ influence

When exposed to a 10 mM NO₃⁻ solution (Fig. S2, Supplementary material) the nZVI was recorded to corrode more rapidly than the control (Fig. S3, Supplementary material), with the emergence of multiple Fe₃O₄γ-Fe₂O₃ peaks after only 2 h and no identifiable Fe⁰ peaks after only 1 d. Magnetite/maghemite remained the dominant phase throughout the experiment. No new peaks emerged after 3 d, and no significant changes were observed after 1 w. At 16 w, Fe₃O₄γ-Fe₂O₃ was the sole phase(s) identified (Fig. 3).

3.3.2. Removal of Cu and Zn by nZVI

The copper uptake mechanism by nano-Fe⁰ has been reported to be via cementation [19], and therefore a source of electrons must be present (Eqs. (1)–(2)):

\[ \text{Cu}^{2+} + \text{Fe}^0 \rightarrow \text{Fe}^{2+} + \text{Cu}^0 \]  

\[ \text{Cu}^{2+} + 2\text{Fe}^0 \rightarrow 2\text{Fe}^{3+} + \text{Cu}^0 \]  

Zinc is removed from solution by surface adsorption and complexation/precipitation, which may include electrostatic interactions and specific surface bonding, with no net electron transfer [19]. The changes in dissolved metal concentrations from the batch systems containing 0.1 and 1.0 mM of Cl⁻, NO₃⁻ and SO₄²⁻ are summarised in Table 2. The changes in solution metal concentrations from the batch systems containing 10 mM of Cl⁻, NO₃⁻ and SO₄²⁻ are summarised in Table 2 and shown in Fig. 4. Considering the extremely low minimum metal concentrations reported in Table 2, it was concluded that solutions containing anions at concentrations up to 10.0 mM had little influence on the initial removal of both Cu and Zn from solution by nZVI as they were recorded at near-zero concentrations after ~1 h. However,

<table>
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<tr>
<th>Anion</th>
<th>Contaminant</th>
<th>Anion conc. (mM)</th>
<th>Minimum metal concentrations (mg L⁻¹)</th>
<th>Remobilisation time (h)</th>
<th>Metal concentrations at 672 h (mg L⁻¹)</th>
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<tr>
<td>Zn</td>
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<td>&gt;1</td>
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<td>&gt;1</td>
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<tr>
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<tr>
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<td>0.199</td>
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<td>ND</td>
<td></td>
<td>&gt;336</td>
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<tr>
<td>Control (anion-free)</td>
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<td>6.799</td>
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<tr>
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when considering metal concentrations at the termination of the experiment in Table 2, it was concluded that differing anionic concentrations exerted a considerable influence on the remobilisation of contaminants into solutions.

For solutions containing Cl– and SO42– at concentrations <1.0 mM, similar aqueous Cu and Zn concentrations to the control were recorded at the termination of the experiment (Cu ~ 6.6 mg L−1, Zn ~ 1.5 mg L−1). However, for solutions containing such anions at 10 mM concentrations (Fig. 4), significant Cu and Zn remobilisation occurred, concurrent with higher concentrations of aqueous Fe. Furthermore, Fe concentrations continued to be elevated until the termination of the experiment, whereas, at lower anionic concentrations, aqueous Fe concentrations were at their highest levels generally for only ~1 h reaction time. This was followed by a reversion to extremely low concentrations after ~2 w (attributed to the oxidation of Fe2+ to the less soluble Fe3+).

Considering remobilisation times in Table 2, for solutions containing Cl– and SO42–, Cu and Zn remobilisation occurred at different times, regardless of anionic concentrations, with Zn generally remobilising before Cu. In contrast, for all solutions containing NO3–, no Cu or Zn remobilisation was recorded, with the concentration of such metals recorded at the end of the experiment (16 w) as near-zero for all solutions regardless of anionic concentration.

3.3.3. Analysis of nZVI aqueous corrosion behaviour

X-ray diffraction plots of the nZVI extracted after 4 w reaction time from the zinc and copper bearing solutions containing 10 mM Cl–, NO3– and SO42– are presented in Fig. S6, Supplementary material; and plots from the 10 mM solutions are presented in Fig. 5. Magnetite/maghemite and γ-FeOOH were identified in the final XRD scans of nanomaterial exposed to the control solution (deionised water only), as well as the 10 mM Cl– and SO42– solutions, whereas a combination of Fe3O4/γ-Fe2O3 and Fe0 were recorded for the nZVI exposed to the solution containing NO3– at 10 mM.

XPS analysis of nZVI extracted from the Cu or Zn solutions after 4 w indicated both Cu and Zn were present on the surface, i.e. sorbed only the nZVI (Fig. 6). Well defined Cu 2p1/2 and 2p3/2 satellite features at ~942 and ~963 indicated the presence of Cu2+ species. [20] The Cu 2p3/2 spectra was fitted with two peaks following the work of Biesinger et al. [21], which indicated the presence of Cu0 (932.5 ± 0.2 eV) and Cu(OH)2 (934.2 ± 0.2 eV). The detection of Cu0 on the surface of nZVI is in agreement with previous work [19]. The Cu2+ ion was present because either it had been directly sorbed to the nZVI from solution (and not chemically reduced) and/or chemically reduced to Cu0 and then reoxidised to Cu2+ (as Cu(OH)2(Zn)). Due to the relatively close distance between the binding energies of ZnP and ZnO species (e.g. ~1021.65 and ~1021.0 eV respectively) mixed systems are difficult to quantify.

Fig. 4. Changes in aqueous metal (top) and iron (bottom) concentration (mg L−1) in solutions containing copper (left) and zinc (right) and 10 mM anions when treated with nZVI, as a function of experimental time (672 h).

Note, different vertical scales. Note, instrument SD values were below 0.01 mg L−1.

Key: Square – Chloride, Circle – Sulphate, Triangle – Nitrate and Cross – Blank.

Fig. 5. X-ray diffraction plots with identified peaks for the range 10–70° 2θ acquired for nZVI extracted from experimental solutions containing Cu (left) and Zn (right) and 10 mM Cl–, NO3–, SO42– and control after 4 w (ascending).

Key: L – γ-FeOOH and M – Fe3O4/γ-Fe2O3.
due to spectral overlap, and therefore the specific Zn phase(s) present could not be determined using XPS.

4. Discussion

4.1. nZVI behaviour when exposed to chloride and sulphate containing solutions

The corrosion rate and reactivity (i.e. the ability to remove Cu and Zn) of nZVI when exposed to Cl⁻ and SO₄²⁻ containing solutions were broadly similar, and therefore these systems will be discussed together. After 16 h exposure, the degree of oxidation of nZVI exposed to 10 mM Cl⁻ and SO₄²⁻ was similar to the nZVI when exposed to the control solution (Fig. S6, Supplementary material). In general this agrees with previous work using hydrogen-reduced nZVI [9], although in such work the nZVI was exposed to a 10 mM SO₄²⁻ solution which resulted in a mixture of corrosion products: schwertmannite (Fe₆₃O₁₆(OH)₄(SO₄)₃·nH₂O) and Fe₃O₄ [17]. The emergence of α-FeOOH (described as an end-product of an iron corrosion series) [17] in the 16 w XRD data was to be considered to be the ultimate corrosion phase of nZVI in the current work.

At high concentrations, Cl⁻ and SO₄²⁻ increased both Cu and Zn remobilisation above control levels. Both Cl⁻ and SO₄²⁻ are regarded as inert with respect to reduction by nZVI (i.e. not redox amenable) [5]. However, these anions have been observed to increase the corrosion rate of Fe⁰ by raising the iron oxide layer dissolution rate [2,4,5], with the breakthrough of this layer leading to a localised corrosion of the underlying metal. Pitting corrosion, an electrochemical oxidation-reduction process, has been observed to occur on the surface of metals coated with a passive film with both Cl⁻ [22] and SO₄²⁻ anions [23]. This process leads to a decrease in solution pH due to hydrolysis [24], which in turn causes an acceleration of the corrosion process. This was observed for the 10 mM Cl⁻ and SO₄²⁻ solutions where increased dissolved iron concentrations (Fig. 4) and low pH levels (Figs. S4 and S5, Supplementary material) were recorded. The greater remobilisation of Cu and Zn into solution from the surface of nZVI was attributed to this pitting corrosion process, with the breakdown of the passive iron surface (the adsorption sites). Being a non-reduced surface complex, Zn is more sensitive to such disturbances at the iron surface, which was shown in the increased remobilisation when compared to Cu.

4.2. Bicarbonate

Bicarbonate is not considered to be reducible by Fe⁰. However, the anion’s role in corrosion is not simple, as both precipitation of FeCO₃ can result in passivation of the nZVI surface [25,26], and adsorbed H₂CO₃ and HCO₃⁻ can react as oxidants providing cathodic reactions that drive nZVI corrosion. In this work, it was evident that HCO₃⁻ caused an acceleration in the nZVI corrosion when compared to the control (Fig. S3, Supplementary material), which has been observed in a previous study with hydrogen-reduced nZVI [9]. This was evidenced by the emergence of the corrosion ‘end-product’ α-FeOOH, and the predominance of this phase after 16 h exposure (Fig. 3). It is thought that an initial film of iron carbonate forms (FeCO₃) readily oxidises (by DO) to Fe₂O₃ [27,28], which explains why iron-carbonate compounds such as siderite, were not detected in any XRD scans; further oxidation of Fe₂O₃/γ-Fe₂O₃ will produce γ-FeOOH [29], but the formation of this phase has been observed to be suppressed or totally eliminated in solutions containing appreciable concentrations of HCO₃⁻ ions, with the formation of α-FeOOH typically occurring instead [30], which was observed in this study. It has been suggested that carbonate anions directly interact with the iron, polymerising the double bands of Fe₂O₃(OH)₂ octahedra common to both γ-FeOOH and α-FeOOH minerals toward a corner sharing arrangement, and thereby to the latter phase [30].

4.3. Nitrate

Unlike the other anions tested herein, NO₃⁻ is a reducible anion (when in the presence of Fe⁰), and is known to greatly impact the ageing behaviour of Fe⁰ [31,32]. In this work, as well as other studies [33,34], NO₃⁻ was shown to transform nZVI into an oxide (Fe₂O₃/γ-Fe₂O₃) rather than a hydroxide compound (e.g. γ-FeOOH/α-FeOOH). Anions like NO₃⁻ are thought to exchange with surface (hydr)oxide ligands to coordinate with oxide–matrix metal ions [4], although there is little agreement on the precise physical and chemical characteristics of this layer; studies have reported highly disordered γ-FeOOH at the surface, with Fe₂O₃ beneath, again in a disordered state [35,36]. The formation of the γ-Fe₂O₃ layer has been proposed to occur via the direct reaction with Fe⁰ or by the oxidation of pre-existing Fe₂O₃ (a common corrosion product of iron) by NO₃⁻. The formation of this layer would account for the lack of remobilisation of metals in the NO₃⁻ solutions (Table 2). This process has been observed to be relatively rapid; the metastable iron phase schwertmannite, detected on hydrogen-reduced nZVI, did not transform to another mineral phase when exposed to a 10 mM NO₃⁻ solution before passivation occurred [9]. This suggests that passivation occurs after the adsorption of the metal contaminant, but before remobilisation can occur (<2 h, Table 2), therefore enmeshing the metals into the surface oxide layer. The low iron concentrations recorded for the NO₃⁻ solutions further confirmed this process (Table 2 and Fig. 4), and therefore ferrous species may not be involved in the Cu or Zn uptake process. Passivation of nZVI surfaces with NO₃⁻ has been reported to occur above concentrations of 3 and 5 mM [5,9]. This work suggests that these figures are too high, and passivation would occur at lower NO₃⁻ concentrations. Greater passivation was observed, however,

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Fig. 6. Typical 2p photoelectron spectra for nZVI removed from 0.3 mM Cu (left) and Zn (right) solutions after 4 w. Note, differing vertical scales.
for the higher concentration NO$_3^-$ solutions (Fig. S6, Supplementary material); Fe$^{0}$ was detected in the XRD data (44.9°–29°) of nZVI removed from the 10 mM NO$_3^-$-Cu solution. The presence of Fe$^{0}$ further implied that the formation of the passivating layer was rapid, because Fe$^{0}$ was generally not detected within the XRD data recorded for nZVI extracted from solutions containing Cu and HCO$_3^-$, Cl$^-$ or SO$_4^{2-}$ after 1 d of aqueous exposure time.

The minimum nitrate concentration of 0.1 mM (6.2 mg L$^{-1}$) which enabled long-term Cu and Zn retention on nZVI was considerably lower than the European Drinking Water Directive’s maximum limit for NO$_3^-$ in water (50 mg L$^{-1}$). However, NO$_3^-$ is a widespread contaminant in groundwater and surface waters [37] and as such care must be taken to ensure that if nZVI and NO$_3^-$ were simultaneously injected into an aquifer for remediation purposes that the process does not result in NO$_3^-$ groundwater contamination.

### 4.4. Cations

Results herein demonstrate that Cu cations were able to oxidise nZVI more rapidly than Zn cations. This can be related to the influence of galvanic coupling, i.e. sacrificial corrosion of nZVI upon formation of a galvanic cell with a redox amenable specie. This was likely to have occurred in solutions containing Cl$^-$ and SO$_4^{2-}$, but not in solutions containing NO$_3^-$ due to its aforementioned passivating influence.

The difference in Cu and Zn remobilisation times (Table 2) was ascribed to the differing uptake mechanism of each metal onto the nZVI. Cu was likely to have been chemically reduced [19] while Zn was likely to have been adsorbed onto the surface, followed by complexation [38]. This suggests that an oxidation step (by the presence of DO) was required for remobilisation of the former species. This was corroborated by both Cu$^{0}$ and Cu$^{2+}$ species being detected at the surface of the nZVI (in the XPS data), whereas Zn was not; Zn can more easily desorb from the nZVI because it is not as redox amenable (not chemically reduced), and therefore may require a constant flow of electrons to remain complexed at the Fe surface.

### 5. Conclusions

For over a decade nZVI has received much attention as a new technology for contaminated land treatment. Despite such interest there still remains fundamental knowledge gaps with regard to the influence of common groundwater anions on both aqueous contaminant uptake onto nZVI and the environmental transformation and fate of nZVI. This study has attempted to bridge this gap by directly comparing the influence of HCO$_3^-$, Cl$^-$, SO$_4^{2-}$ and NO$_3^-$ on the aqueous corrosion of nZVI and associated removal efficacy of aqueous Cu and Zn. The following can be concluded:

i) The corrosion rate of nZVI was recorded to follow the order (from most reactive to most passive) of: HCO$_3^-$ > Cl$^-$ > SO$_4^{2-}$ > NO$_3^-$ when nZVI was exposed for 16 w to a 10 mM HCO$_3^-$ solution the main corrosion product phase was α-FeOOH. In contrast, Fe$_2$O$_3$γ–Fe$_2$O$_3$ was recorded as the main nZVI corrosion product phase for solutions containing Cl$^-$ or SO$_4^{2-}$, with a minor contribution from γ-FeOOH and α-FeOOH. The main nZVI corrosion product phase when exposed to NO$_3^-$ was Fe$_2$O$_3$γ–Fe$_2$O$_3$.

ii) The ability of nZVI to initially remove Cu and Zn from solution was not influenced by common groundwater anions (near-total removal in all systems was recorded). However, the long-term retention of Cu and Zn by nZVI was strongly influenced by the presence of Cl$^-$, SO$_4^{2-}$ and NO$_3^-$, which phenomena was recorded to only occur when anions were present at 10 mM concentrations, with Cu and Zn largely unaffected when the anions were present at concentrations below 1.0 mM. Cu and Zn desorption from the nZVI was recorded to be enhanced by the presence of both SO$_4^{2-}$ and Cl$^-$ at 10 mM. This was ascribed to pitting corrosion (driven by the presence of SO$_4^{2-}$ and Cl$^-$) which acts to destabilise the iron surface and allow sorbed metals to be released back into solution. Very little desorption of Cu and Zn was recorded for solutions containing NO$_3^-$ due to passivation of the nZVI surface.

Overall the results demonstrate that HCO$_3^-$, Cl$^-$, SO$_4^{2-}$ and NO$_3^-$ have considerable influence on both nZVI aqueous corrosion behaviour and heavy metal retention. Whilst Cl$^-$ and SO$_4^{2-}$-bearing solutions were recorded to enhance nZVI corrosion rate and also cause Cu and Zn desorption, the opposite was recorded for solutions containing NO$_3^-$, which is therefore suggested that nitrate could be utilised as a complimentary chemical to prevent metal desorption from nZVI and thus improve its long-term performance for in situ water treatment applications.

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### Appendix A. Supplementary data

Supplementary data associated with this article can be found, in the online version, at [http://dx.doi.org/10.1016/j.jece.2017.01.038](http://dx.doi.org/10.1016/j.jece.2017.01.038).

### References


