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Removal of Pb(II) from aqueous solution by a zeolite–nanoscale zero-valent iron composite

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1. Introduction

Heavy metals are problematic for ecosystems because of their toxicity and most heavy metals can be highly toxic even at very low concentrations. Among these, Pb is commonly used in several industries and in some locations large amounts of wastewaters containing high concentrations of Pb ions have been released. Lead directly or indirectly reaches surface and ground water and becomes biomagnified in biotic communities. Lead primarily accumulates in muscles, bones, kidneys, and brain tissues and can cause anemia, nervous system disorders, and kidney diseases [1]. Conventional ion exchange, filtration, adsorption, chemical precipitation, and reverse osmosis are being used to remove metals from water [2]. Among these methods, adsorption is a highly efficient and economical removal technique [3].
Permeable reactive barriers (PRBs) are a cost-effective in situ technology for removing a wide array of contaminants from ground water. Optimization of reactive materials remains a major challenge in developing effective PRB technology. Zero-valent iron (ZVI) is being used to remove heavy metals from ground water but low reactivity and handling difficulties have reduced its application in PRBs [4]. Alternatively, nanoscale zero-valent iron (nZVI) has shown good potential to remove metals and other aqueous pollutants. Its physicochemical properties and reductive capacity can facilitate rapid decontamination of polluted water [5, 6]. Unfortunately nZVI often forms aggregates, which decreases efficiency by reducing surface area [7] and producing a less negative oxidation–reduction potential [8]. To resolve this problem, various immobilization techniques are being developed for nZVI stabilization. Wei et al. [7] stabilized nZVI with biodegradable surfactant for effective removal of vinyl chloride and 1,2-dichloroethane from water. Zhang et al. [9] prepared nZVI with pillared clay as a stabilizer for nitrate removal from water. Liu et al. [10] used chitosan to reduce nZVI aggregation and Calabrò et al. [11] prepared nZVI with a pumice granular mixture to remove nickel ions from water.

Zeolites are microporous, aluminosilicate minerals commonly used as adsorbents for several pollutants. Natural zeolites have a high sorption capacity for inorganic pollutants, including heavy metals and ammonium [12]. Basaldella et al. [13] used NaA zeolites to remove Cr from water. Cs and Sr were removed from aqueous solution using zeolite A [14]. Cincotti et al. [15] reported preferential removal of Pb over Cu, Cd, and Zn by a Sardinian natural zeolite and Panayotova and Velikov [16] found that Pb(II) was effectively immobilized by Bulgarian natural zeolite. More recently, Yang et al. [17] showed that NKF-6 zeolite effectively removed Pb(II) from a large volume of water. Zeolites have proven effective for environmental applications such as in PRBs for controlling the spread of cation-contaminated groundwater [18]. However, only limited attempts have been made to stabilize nZVI with zeolites for removal of pollutants from water [19]. Lee et al. [19] used a zero-valent iron zeolite material (ZanF) for nitrate reduction without ammonium release under unbuffered pH. ZanF removed the ammonia to below detection limits via adsorption, whereas ZVI alone did not remove it to any significant extent.

The objectives of the present study are to (i) synthesize and characterize a zeolite–nZVI composite (Z–nZVI) and (ii) assess its reductive capacity for Pb(II) release.

2. Materials and methods

2.1. Materials and chemicals

Naturally occurring zeolite was obtained from Alfa Aesar, A Johnson Matthey Co., Seoul, South Korea. The zeolite was composed of Al₂(SiO₄)₃, Na, Ca, K, and H₂O and had a Mohs hardness of 3.5–5.5. The cation exchange capacity (CEC) of the zeolite was 105.38 cmol/kg, within the typical range for natural zeolites [12]. After drying at 80 °C overnight, the zeolite was ground and sieved with a 100 mesh screen before use. Ethylene diaminetetraacetic acid (EDTA; DAE JUNG, Siheung, Korea) was >99% pure. All other chemicals were analytical grade. Nanopure water (conductivity = 18 μΩ/m, TOC < 3 μg/L; Barnstead, Waltham, MA, USA) was used to prepare all reagents. A Pb stock solution was prepared by dissolving 1.60 g Pb(NO₃)₂ in 100 mL of degassed water and working concentrations were prepared by diluting the stock solution.

2.2. Preparation of the composite

The Z–nZVI composite was prepared according to Wang et al. [21]. Briefly, 1 g of FeSO₄·7H₂O and 0.5 g of natural zeolite were mixed in 250 mL of degassed nanopure water. The pH of the solution was adjusted to 4 with 1 M HNO₃. The mixture was treated with ultrasound for 10 min, and then stirred vigorously at ambient temperature for 30 min. To ensure efficient reduction of Fe(II), 25 mL of 1 M K BH₄ solution was added at 30 drops/min while stirring. The reduction reaction is as follows:

\[
\text{Fe}^{2+} + 2\text{BH}_4^- + 6\text{H}_2\text{O} \rightarrow \text{Fe}^0 + 2\text{B(OH)}_3^- + 7\text{H}_2↑
\]

After incubation, the black solids were separated from the solution using a vacuum filtration flask (0.45 μm membrane filter), washed several times with degassed water to remove residual sulfate, then vacuum-dried.

2.3. Characterization of the composite

Field emission scanning electron microscopy (FE-SEM; Hitachi S-4700, Tokyo, Japan) was used to view the morphology and surface characteristics of the nZVI and zeolite. The characteristics of the Z–nZVI composite were obtained using biological transmission electron microscopy (Bio-TEM; Hitachi H-7650, Tokyo, Japan) and energy-dispersive X-ray spectrometry (EDS) were obtained using FE-SEM. Surface areas of the zeolite, nZVI, and Z–nZVI composite were measured by N₂ adsorption using a Micromeritics ASAP (Accelerated Surface Area and Porosimetry) 2020 analyzer (BELSORP-MINI, BEL Japan, Inc., Osaka, Japan) [6]. Infrared spectra of the zeolite, nZVI, and Z–nZVI composite powders were obtained in KBr pellets on a Perkin–Elmer Fourier transform infrared (FTIR) spectrophotometer (Irvine, CA, USA) in the diffuse reflectance mode at a resolution of 4 cm⁻¹.

2.4. Pb(II) removal and release

The procedures of Zhang et al. [22] were used to determine the effects of initial pH (2–6), temperature (5–60 °C), and Pb(II) concentration (100, 250, 500, and 1000 mg/L) on adsorption to the Z–nZVI composite. The initial pH of the solutions was adjusted using 0.1 M HCl or 0.1 M NaOH but was not controlled during the experiments. Briefly, 0.1 g of the composite was mixed with 100 mL of Pb(II) solution (100 mg/L) and placed on a rotary shaker at 180 rpm and room temperature. Samples were collected periodically up to 140 min and filtered using a 0.45 μm syringe filter. Pb(II) concentration in the filtrate was determined by ICP-AES (Inductively Coupled Plasma, Leeman Labs, Inc., Hudson, NH, USA). Zeolite was used as the control for this experiment.

A sequential extraction procedure was applied to the Pb(II)-loaded Z–nZVI composite to determine Pb(II) availability, following the general procedures of Basta and Gradwohl [23] and Castaldi et al. [24]. To extract readily available Pb(II), Z–nZVI composite (1 g) containing 1.3 mg Pb(II) was shaken with 25 mL of nanopure water (pH 6.8) for 2 h at room temperature (26 °C). The composite was then sequentially extracted with 25 mL of 0.1 M Ca(NO₃)₂ (pH 7.8) to remove exchangeable Pb(II), followed by 25 mL of 0.1 M EDTA (pH 8.0) to remove the more tightly bound Pb(II) or Pb hydroxide complexes precipitated on active sites that were not readily bioavailable [23–25]. After the extractions, the composite was dried overnight at 105 °C and digested with 0.1 M HNO₃ and 0.1 M HCl to recover Pb(II) and other non-exchangeable Pb (likely present as Pb oxides or mixed Pb–Fe oxides). After each extraction the composites were...
centrifuged (6000 rpm for 10 min) and filtered to separate the solution and solid phases [23].

2.5. X-ray diffraction

To determine the nature of the Pb associated with the composite, X-ray diffractograms (XRD) of dried Z–nZVI were obtained after shaking with Pb solution. A Cu Kα incident beam ($\lambda = 0.1546$ nm) was used, monochromated by a nickel filtering wave at a tube voltage of 40 kV and current of 40 mA (Philips X’Pert Pro MPD, Eindhoven, Netherlands). Scanning was at a 2$^\theta$ range of 30–70° at 0.04 deg/min with a time constant of 2 s.

3. Results and discussion

3.1. Characterization of the composite

Typical SEM images of nZVI and zeolite and TEM images of the Z–nZVI composite are shown in Figure 1a–c. As previously reported, nZVI particles become aggregated (attributable to van der Waals and magnetic forces) [7, 26] and the aggregation decreases nZVI reactivity and mobility [27]. Stabilizing supports such as zeolite have been used to prevent aggregation [28]. In our Z–nZVI composite, the zeolite decreased aggregation and the nZVI was present in chain-like structures. EDS further confirmed the presence of Fe in the composite (Figure 1d). The mean surface area of the composite was 80.37 m$^2$/g, compared to 12.25 m$^2$/g for nZVI and 1.03 m$^2$/g for the zeolite alone. The increased surface area of the composite is likely due to non-aggregation of the nZVI particles.

The FTIR spectrum of the Z–nZVI composite supports nZVI loading onto the zeolite. Broad bands at 3400–3600 cm$^{-1}$ in zeolite and the composite (Figure 2b and c) result from O–H stretching, likely due to $\text{H}_2\text{O}$ and M–OH, while the band at 1650 cm$^{-1}$ can be attributed to O–H bending [29]. The peak at 3500 cm$^{-1}$ and those at 3200, 3100, 3000, 2550 and 2050 cm$^{-1}$ in nZVI can be attributed to the stretching vibrations of O–H groups. Most of these bands disappeared in the composite, indicating loss of water molecules [30, 31]. Bands at 1200–900 cm$^{-1}$ result from $\text{SiO}_2$ and $\text{AlO}_2$ stretching in the zeolite, with bending modes at ca. 740 and 689 cm$^{-1}$ [32, 33]. Major weakening of the zeolite band at 1000 cm$^{-1}$ in the composite and band shifts in this region suggest H-bond breaking due to the presence of Fe on the $\text{SiO}_2$ and $\text{AlO}_2$ surfaces of zeolite [33]. Strong bands at $<$900 cm$^{-1}$ in the nZVI alone (Figure 2a), attributable in part to iron oxides on the surface [31, 34], are weaker in the composite, indicating less oxidation of zeolite-supported Fe$^0$. The zeolite support may have reduced Fe (oxy)hydroxide formation, similar to the effect of montmorillonite-supported nZVI [35]. Bands at ~1300 and ~1100 cm$^{-1}$ in the nZVI can be attributed to ethanol used in preparing the sample, but may also include bands associated with sulfate green rust [Fe$^{\text{II}}$Fe$^{\text{III}}$(OH)$_{12}$[SO$_4$]$_2$3H$_2$O] [36, 37] and lepidocrocite ($\gamma$-FeOOH) [34] formation on some Fe$^0$ surfaces.

3.2. Removal of Pb(II) from water

Figure 3 shows solution concentrations of Pb(II) as a function of reaction time for 0.1 g of Z–nZVI composite or zeolite in 100 mL of solution containing 100 mg Pb(II)/L at 35 °C. Adjusting the initial pH to 4 dissolved the passivating Fe (oxy)hydroxide layer on nZVI surfaces [38]; the solution pH increased to 7.7 during equilibration due to reaction of nZVI with water [39]. Results indicate that the composite effectively removed 96.2% of the Pb from aqueous solution (96.2 mg/g) within 140 min, while the zeolite alone only removed 39.1% (39.1 mg/g). The enhanced effectiveness of the Z–nZVI composite for Pb(II) removal is likely due to its much larger specific surface area than that of zeolite alone. The zeolite supporting material prevented aggregation of nZVI, thereby providing more surface area for Pb(II) sorption [31]. Results are consistent with previous studies reporting adsorption of Pb(II) by kaolinite-supported nZVI, and Cr(VI) and Pb(II) adsorption by resin-supported nZVI [31, 40].
3.2.1. Effect of Pb(II) concentration

The effect of initial Pb(II) concentration (100–1000 mg/L) on removal efficiency was investigated by shaking 0.1 g of the composite in 100 mL of solution containing 100 mg Pb(II)/L for 140 min at pH 4 and 35 °C. Removal efficiency varied with initial concentration (Figure 4). At the lower concentration (100 mg/L), 99.2% of the Pb(II) was removed by the Z–nZVI composite (99.2 mg/g). The decrease in removal efficiency to 80.6% at the higher concentration (1000 mg/L) suggests that the capacity of the Z–nZVI is about 806 mg Pb(II)/g, which was exceeded under the conditions of the experiment, as observed for removal of Pb(II), Cu(II), and Zn(II) by natural zeolite [41] and Cr(VI) ions by a bentonite–nZVI composite [42].

3.2.2. Effect of initial pH

Solution pH can have a significant influence on the adsorption of heavy metals, due to metal speciation, surface charge, and functional group chemistry of the adsorbent [43]. Hence, 0.1 g of the Z–nZVI composite was mixed with 100 mL of solution containing 100 mg Pb(II)/L at an initial pH of 2–6 (26 ± 2 °C). The pH of the solutions was adjusted before adding the Z–nZVI composite, but increased from 2 to 6.1, 3 to 7.4, 4 to 7.7, 5 to 8.2 and 6 to 7.8 during the experiment, primarily from oxidation of Fe(0) (and Fe(II)) by water [39]. Varying the initial pH had a small effect on Pb(II) removal efficiency (Figure 5); removal ranged from 99.9% when the initial pH was 4 to 93.5% when it was 6. The difference in pH would have a minimal effect on the surface charge of zeolite [44]. Although Pb²⁺ ions predominate in solution at acidic
pH, competition from protons decreases removal at an initial pH of 2 [22]. Conversely, when the initial pH was 6 the presence and adsorption of Pb(OH)$^+$ may have prevented Pb$^{2+}$ diffusion to some sites within the porous zeolite structure [41]. The greater Fe (oxy)hydroxide coating on nZVI surfaces at an initial pH of 6 would also decrease reactivity, reflected in a smaller pH change during the experiment. Our results suggest that rapid diffusion of Pb$^{2+}$ into the Z-nZVI matrix and adsorption were optimized by adjusting the initial pH to 4, and were followed by reduction to Pb$^0$ by Fe$^0$. The more acidic solution pH facilitates these processes through dissolution of the passivating Fe (oxy)hydroxide layer on nZVI surfaces [38]. While aggregation of nZVI may increase near its effective point of zero charge, which likely ranged from 6–8 due to surficial Fe (oxy)hydroxides, most of the Fe$^0$ is immobilized on zeolite in the Z–nZVI composite.

### 3.2.3. Effect of temperature

Temperature is an important factor affecting adsorption and would be generally expected to increase with decreasing temperature due to the exothermicity of cations for an adsorbent surface. Temperature had a relatively small effect on Pb removal by the Z–nZVI composite, which ranged from 99.8% at 60 °C to 94.6% at 5 °C (Figure 6). More efficient removal at higher temperatures is likely due to desolvation of Pb cations [17] and more rapid diffusion into the internal pores of the composite particles. Results are consistent with the greater adsorption of Pb(II) on NKF-6 zeolite [17] and Cr(VI) on a bentonite-nZVI composite [42] with increasing temperature.

### 3.2.4. X-ray diffraction

XRD patterns of the Z–nZVI composite were recorded before and after shaking with the aqueous solution alone (Figures 7a and b, respectively) or with the Pb solution (Figure 7c). Peak 1 (and 4) at 2θ ~ 32 likely arises from SiO$_2$ associated with the natural zeolite [45], while that at 2θ = 45 (2 and 12) is characteristic of FeO [33; JCPDS00-006-0696]. Fe(II) adsorbed to the zeolite was likely reduced to Fe$^0$ and immobilized on the surface, as described by Lee et al. [19]. Peak 3 (2θ = 50) is likely due to maghemite ($\gamma$-Fe$_2$O$_3$) on some of the Fe$^0$ [46]. Peaks 5–10, appearing in Z-nZVI after shaking with aqueous solution, can be attributed to the formation of iron oxides, primarily magnetite (Fe$_3$O$_4$), maghemite, and lepidocrocite from Fe$^0$ oxidation [31]. The peaks at 2θ ~ 35 (11) and ~ 62 (14)

in the Z–nZVI composite after exposure to Pb(II) solution (Figure 7c) are attributed to Pb$^0$ [31, 40], while that at 2θ ~ 57 (13) is likely an iron oxide. The XRD analyses support formation and immobilization of Fe$^0$, as well as sorption and reduction of Pb(II) to Pb$^0$, on the composite.

### 3.3. Availability of Pb removed by the composite

The Pb(II)-loaded composite was sequentially shaken with extractant solutions of increasing removal capacity to determine the availability of Pb associated with the composite. Lead readily extractable with water comprises the most soluble and bioavailable fraction. That fraction was less than 0.5% of the adsorbed Pb(II) (Table 1). The fraction extractable with Ca(NO$_3$)$_2$ comprised exchangeable Pb, which was about 2.3% of the Pb initially removed by the Z–nZVI composite. In contrast, the fraction extracted with EDTA, consid-

<table>
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<tr>
<th>Extractant</th>
<th>Pb (μg)</th>
<th>% of total</th>
</tr>
</thead>
<tbody>
<tr>
<td>Initial amount in the Z–nZVI composite</td>
<td>1303.25 ± 0.03</td>
<td>100.0</td>
</tr>
<tr>
<td>H$_2$O</td>
<td>5.30 ± 0.08</td>
<td>0.4</td>
</tr>
<tr>
<td>Ca(NO$_3$)$_2$ (0.1 M)</td>
<td>29.45 ± 6.73</td>
<td>2.3</td>
</tr>
<tr>
<td>EDTA (0.1 M)</td>
<td>1074.50 ± 6.18</td>
<td>82.5</td>
</tr>
<tr>
<td>Digestion of Z–nZVI with HNO$_3$/HCl</td>
<td>193.05 ± 57.28</td>
<td>14.8</td>
</tr>
</tbody>
</table>
erated as not readily bioavailable [23–25], was 82.5% of the adsorbed Pb(II). The residual fraction, which is not expected to be readily released under natural conditions, comprised 14.8% of the Pb(II) associated with the composite. Aside from Pb(II) resulting from Pb(II) reduction, this fraction may include some Pb replaced for Al within the zeolite lattice [24].

The low quantity of Pb(II) recovered in the water-soluble and Ca(NO3)2-extractable fractions (Table 1) indicates low bioavailability of the Pb removed by the Z-nZVI composite. These fractions likely consist of Pb2+ electrostatically adsorbed to external surfaces of the composite [24]. The large fraction of Pb(II) extracted with EDTA likely consists of more strongly bound Pb(II) and precipitated lead hydroxide complexes on active sites within the zeolite-based matrix of the composite [41, 44]. EDTA is known to be highly effective for extracting lead from soil and the Pb-EDTA complex has high stability at pH 8 [47], the pH of the EDTA solution in our experiment. Our results suggest that a large fraction of the Pb(II) removed by the Z-nZVI was incorporated into the internal matrix of the composite. We postulate that the nZVI within the composite sequestered the Pb(II) and gradually reduced it to Pb0, as described for adsorption and reduction of Ni(II) by nZVI [48]. Once the metal becomes incorporated within the composite structure, it remains essentially insoluble and non-exchangeable [24].

4. Conclusions

Zeolite was an effective dispersant and stabilizer of nZVI in a composite support system, reducing aggregation and increasing specific surface area. Batch experiments indicated that the Z-nZVI composite was superior to zeolite in removing Pb(II) from aqueous solution. XRD confirmed that the composite adsorbed the Pb(II) ions and subsequently reduced some of them to Pb0. Because zeolite is a stable and low-cost mineral, the zeolite-nZVI composite is an efficient and promising reactive material for PRBs. Further studies are needed to assess the potential of the material to remove other metals and organic pollutants.

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