Electrochemical denitrification of simulated groundwater

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Abstract

Electrochemical denitrification of groundwater was studied with an objective to maximize nitrate transformation to nitrogen gas. Aluminum, graphite, iron and titanium were selected as electrode materials. While aluminum, iron and titanium electrodes showed 70–97% nitrate reduction, with graphite electrode the removal was only 8%. Nitrate was transformed to ammonia with iron and aluminum electrodes but with titanium electrodes nitrogen was apparently the major end product. Iron electrodes exhibited the maximum reducing condition (ORP = −463 mV) and titanium showed the minimum (ORP = −206 mV). Nitrate reduction with titanium electrodes was retarded in the presence of chloride ions possibly due to formation of hypochlorite ions. The first-order nitrate transformation rate constant with respect to time decreased with decrease in current density. However, when the rate constant was expressed with respect to charge passed (Coulomb/l) it was nearly same under different experimental conditions (current density and pH). The study indicates that the process might be suitable for denitrification of drinking water.

Keywords: Electrochemical reduction; Nitrate; Ammonia; Nitrogen; Iron; Titanium

1. Introduction

Recent reports reveal that nitrate contamination of water resources is an ubiquitous environmental problem. US geological survey in its recent studies in US has reported widespread occurrence of nitrate in ground water (Squillace et al., 2002). The ground water nitrate concentration is also high in India as well as European countries. In some parts of India exceptionally high ground water concentrations of nitrate (up to 1800 mg/l nitrate-N) have been found (Majumdar and Gupta, 2000; Mehta et al., 1990). The reason for increase in nitrate levels in groundwater has been mainly attributed to profligate use of nitrogenous fertilizers (Mathur and Kumar, 1990).

Nitrate-contaminated water above the permissible drinking water limit is toxic to human health, specifically to children (Gupta et al., 2000; Majumdar and Gupta, 2000; Shrimali and Singh, 2001) and the drinking water standard have been promulgated as 10 mg/l as NO\textsubscript{3}-N.

Properties like high solubility, less ability for precipitation and adsorption makes nitrate removal a difficult task. Physicochemical processes such as ion exchange, reverse osmosis and electrodialysis are effective for the purpose (Kapoor and Viraraghavan, 1997). However, high installation and maintenance costs, brine generation and sensitivity of membranes are some of the limitations of membrane processes. Nitrate ion has high chemical stability at low concentrations and chemical reduction of nitrate requires special conditions such as catalysts and high temperature and pressure (Fanning, 2000). Thus bio-electrochemical methods have been evaluated for the purpose (Flora et al., 1998). Bacterial contamination,
increase of the chlorine demand of treated water and presence of residual organics in treated water undermines the use of biological processes for drinking water treatment. In addition to high sulfate production, low DO concentrations as well as foul smell of the treated water restrict the application of sulfur-based autotrophic denitrification process (Kapoor and Viraraghavan, 1997). Hence, there is a need to study and develop a cost effective and robust process for drinking water denitrification. Electrochemical reduction might be an alternative approach to remove nitrate from ground water. Electrochemical processes can be operated with simple reactor configuration and are amicable to modification and automation. The treated water does not require any post-treatment because there is no microbial contamination. Unlike biological process the process does not require any startup period. The process being selective for reduction of nitrate and as extra chemicals are not being added, thereby major composition of raw water can be retained. The mechanism of electrolytic reduction of nitrate is complex. Among the products formed ammonia is found as major species formed via reduction of nitrite intermediate (Chebotareva and Nyokong, 1997; De et al., 2000a, b). There are few reports on reduction of nitrate to nitrogen (Gender et al., 1996; Li et al., 1988b). Besides these two major products there is possibility of formation of other products such as N2O, NO, and NH2OH. The products formed during electrolysis depend on experimental conditions such as pH, cell configuration, coexisting ions and choice of electrode materials (Li et al., 1988a, b, Peel et al., 2003). In general, applications of the electrochemical process for denitrification are limited due to generation of ammonia.

Researchers have focused on the electrochemical treatment of water either in alkaline medium or in simulated conditions containing high concentration of nitrate basically oriented towards treatment of nuclear waste (Bockris and Kim, 1996; Gender et al., 1996). Present work evaluates electrochemical nitrate removal for groundwater treatment. From the previous discussion, it is obvious that electrochemical process can be applied for denitrification if the end product is nitrogen instead of ammonia. Hence the focus of the present study was to conduct experiments to maximize transformation of nitrate to nitrogen. Aluminum, graphite, iron and titanium have been used as electrode materials. Experiments were conducted in an undivided electrochemical cell at low current density.

2. Materials and methods

2.1. Preparation of simulated ground water

Simulated ground water (SGW) with varying nitrate concentrations (100–200 mg/l NO3⁻-N with 1:1 NaNO3 and KNO3) were prepared in distilled water (Hoek et al., 1988) to be used as electrolyte. All chemicals were analytical reagent grade and were used without any further purification. All glassware were cleaned with 1 N sulfuric acid, rinsed with distilled water and dried in oven before being used. The composition of simulated ground water used for electrolysis is given in Table 1 and pH of the simulated ground water was 8 ± 0.2.

2.2. Procedure

Electrochemical setup was designed as shown in Fig. 1. The constituents included a rectifier (Power electronics, 0–20 V), ammeter, voltmeter and the electrochemical cell. Rectifier was used to pass constant electric current and maintain constant current density. Plexiglass container of 1000 ml was used as electrochemical cell unless otherwise stated. Magnetic stirrer was used for continuous mixing of the electrolyte. pH was monitored (Control Dynamics pH meter) and maintained by addition of concentrated HCl or NaOH. Current and voltage were monitored with ammeter and voltmeter, respectively. Temperature was maintained at 30°C with the help of a water bath.

<table>
<thead>
<tr>
<th>Parameter</th>
<th>Value</th>
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<tbody>
<tr>
<td>Na⁺</td>
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<tr>
<td>K⁺</td>
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<tr>
<td>SO₄²⁻</td>
<td>30 mg/l</td>
</tr>
<tr>
<td>Cl⁻</td>
<td>30 mg/l</td>
</tr>
<tr>
<td>HCO₃⁻</td>
<td>100 mg/l</td>
</tr>
<tr>
<td>NO₃⁻</td>
<td>100 mg/l</td>
</tr>
<tr>
<td>pH</td>
<td>8 ± 0.2</td>
</tr>
<tr>
<td>Conductivity</td>
<td>1.57 mS</td>
</tr>
</tbody>
</table>

Table 1 Composition of simulated ground water

Fig. 1. Schematic representation of electrochemical setup used for nitrate removal.
Aluminum, graphite, iron and titanium plates of appropriate sizes were used as electrodes. All electrodes were made to maintain an immersed electrode face area of 16 cm² inside the cell unless otherwise stated. The inter electrode distance was maintained constant (0.6 cm) in all experiments. Electrodes were cleaned with sand paper and washed with 1 N sulfuric acid and rinsed with water before use to remove impurities and oxide layer. Effects of two common ions present in the ground water such as chloride and sulfate ion on nitrate reduction was observed with titanium electrodes. The simulated ground water was added 250 mg/l sodium chloride for effect of chloride and 250 mg/l sodium sulfate for effect of sulfate. Duplicate runs were carried out for each experiment. Samples were taken out at different time intervals and filtered through filter paper and the filtrate was used for analysis. The results obtained were used to obtain kinetic constants.

2.3. Analytical methods

All analyses were done according to Standard methods (APHA, AWWA, WPCF, 1976, 1998) and all reagents were of analytical reagent grade. Nitrate was estimated by both colorimetric brucine sulfate method (APHA et al., 1976) using spectrophotometer (Spectronic 20 Genesy, USA), and by UV method using spectrophotometer (UV-260 Schiamdzu, Japan). Standard curve for colorimetric method was plotted by preparing nitrate-N standards in the range of 0–10 ppm. Standard curve for UV-method was prepared by directly taking the standard nitrate solution in the range of 0–11 ppm (APHA, AWWA, WPCF, 1998). Nitrite was colorimetrically estimated by sulfanilamide method (APHA, AWWA, WPCF, 1998). Ammonia was colorimetrically estimated by Nesslerization method (APHA, AWWA, WPCF, 1998). Electrode corrosion was measured by taking the weight of the electrode before and after experiments. Dissolved iron and aluminum content of the filtered samples were estimated colorimetrically after filtering the sample over filter paper at the end of the experiments. For iron estimation 1, 10-phenanthroline method was used (APHA, AWWA, WPCF, 1998). Dissolved aluminum estimation was done by eriochrome cyanine method (APHA, AWWA, WPCF, 1998). Sample from the cell was taken out and ORP was measured immediately with Orion pocket ORP meter (Model 108, Orion Research, USA).

3. Results and discussion

3.1. Electrode screening

Nitrate transformation to nitrite, nitrogen (gas) or ammonia is a reductive reaction. The reductive environment (ORP) in the electrochemical cell is mainly influenced by the electrode material. Hence, the material which would produce a reducing environment just sufficient to transform nitrate to nitrogen (gas) would be the preferred material. Gender et al. (1996) reported good denitrification of alkaline nuclear waste with graphite and iron electrodes. Accordingly, graphite, iron, aluminum and titanium were selected for the study. Initial experiments were conducted to prescreen electrode materials based on nitrate reduction efficiency. Experiments with graphite electrodes revealed its inefficiency to transform nitrate from the electrolyte. With graphite electrodes only 8% nitrate removal was achieved after 9 h under the experimental conditions employed (current density: 140 mA/cm² and 300 ml liquid volume). But nitrate transformation up to 80% was noticed after 5 h with aluminum, iron and titanium electrodes under similar experimental conditions. The higher removal with aluminum, iron and titanium electrodes may be due to their greater electron donating capacity. Graphite is a non-metal and a poor electron donor compared to metals. Cyclicvoltametric studies with graphite electrodes indicated low nitrate reduction efficiency (Bouzek et al., 2001). Gender et al. (1996) reported good nitrate reduction efficiencies with graphite (up to 75%) with divided cell containing strong alkaline solutions of nitrate and nitrite. They described the improved removal efficiency in divided cells was due to the elimination of anodic interference or anodic oxidation of nitrite (which acts as an intermediate during nitrate reduction). The observed low nitrate removal with graphite in the present study might be due to effects of ions presents in simulated water as well as anodic oxidation of nitrite in undivided conditions. Thus, one compartment cell might not be suitable with graphite electrodes.

3.2. Iron electrodes

Fig. 2 shows the variation of total nitrogen (the sum of nitrate-N, nitrite-N and ammonia-N), nitrate-N, nitrite-N and ammonia-N during electrolysis with iron electrodes at pH 7 and 9. The nitrate-N transformation at pH 7 and 9 after 5 h of elapsed time is nearly 80%. At both cases the formation of ammonia is very high and very less concentration of nitrite has been detected. There is no decrease in total nitrogen concentration at pH 7 where as at pH 9 the total nitrogen concentration decreases during operation. The possible reason for decrease in total nitrogen concentration at pH 9 can be explained by considering the dissociation of ammonium ion in water which is a function of pH (Metcalf and Eddy Inc, 2003). The distribution of ammonium ion and free ammonia at pH 7 and 9 are as follows: At pH 7 — NH₄⁺: 99.44% and free ammonia (NH₃): 0.56%; at pH 9 — NH₄⁺: 64% and free ammonia (NH₃): 36%.
Thus stripping of ammonia at pH 7 would be quite less and thereby the ammonia formed during the process would remain in solution where as at pH 9 stripping of ammonia would occur (due to high free ammonia concentration). Ammonia stripping might be the cause for decrease in total nitrogen concentration at pH 9.

The pathway of nitrate transformation is expected to be:

\[
\text{Oxidation state (N): } \quad \text{NO}_3^- \rightarrow \text{NO}_2^- \rightarrow \text{NO} \rightarrow \text{N}_2\text{O} \rightarrow \text{N}_2 \rightarrow \text{NH}_3
\]

Thereby accumulation of intermediate products is also expected. But the quantity of nitrite accumulated was very less at both pH 7 and 9 with iron electrodes. It seems that the conversion of nitrite to ammonia was very fast. It is clear from total nitrogen value (Fig. 2(a)) that nitrate gets converted to ammonia in the reactor containing iron electrodes. Formation of ammonia is a cause of concern because total nitrogen remains unaltered in the milieu. The total dissolved iron concentration in the treated water was 1.67mg/l at pH 9. Much higher values of iron than the theoretical value (<0.05mg/l) might be due to the presence of Fe\(^{2+}\) in water. High dissolved iron concentration though not toxic for human use, its presence creates color problem of the treated water and thus can be taken as a limitation of using these electrodes. Anodic corrosion rates of (which can be termed as sludge generation rate) Fe at pH 7 and 9 were found to be 10.02 and 10.1 moles Fe/mole of NO\(_3^-\)N reduced, respectively.

3.3. Aluminum electrodes

Fig. 3 shows transformation of nitrate with aluminum electrodes at pH 7 and 9. It is evident from the figure that major fraction of nitrate gets transformed to ammonia with aluminum electrodes. The amount of nitrite accumulation is higher compared to iron.
electrodes (Figs. 2(a) and (b)). Plausible explanation for accumulation of nitrite may be the rate of conversion of nitrite to ammonia is less than that observed with iron electrodes and the reducing environment inside the cell containing aluminum electrodes is also less than the cell containing iron electrodes. Decrease in total nitrogen concentration at pH 7 with aluminum electrodes indicates the probable formation of other products because loss of ammonia by stripping would be negligible as the free ammonia at pH 7 is very less. This decrease in total nitrogen level cannot be attributed to ammonia stripping because no decrease in total nitrogen level has been observed with iron electrodes at pH 7 (Fig. 2(a)). The dissolved aluminum concentrations were found 0.048 and 0.032 mg/l at pH 7 and 9, respectively. The anodic corrosion rates of Al at pH 7 and 9 were 14.98 and 16.1 moles Al/mole of NO3-N reduced, respectively.

3.4. Titanium electrodes

Fig. 4 presents nitrate transformation in the reactor containing titanium electrodes. It is evident from Figs. 4(a) and (b) that generation of ammonia with titanium electrodes is considerably less. The formation of nitrite with Ti electrodes is also more than other electrodes. Generation of ammonia has been qualitatively verified by placing concentrated HCl rod over the reactors. The formation of ammonium chloride was observed by dense white fumes on the cell containing iron and aluminum electrodes, but the same effect was not observed on the cell containing titanium electrodes. There was decrease in total nitrogen concentrations at pH 7 and 9. This indicates that ammonia is not the major transformed product at both conditions. It is also evident from the figures that accumulation of higher concentration of nitrite occurred with titanium electrodes. Thus the rate of transformation of nitrite to ammonia as well as the reducing conditions inside the reactor seems to be less with titanium electrodes. On the basis of low ammonia production, further studies in the presence of chloride and sulfate ions were conducted with titanium electrodes to observe the effect of these ions on nitrate transformation. Table 2 shows the nitrate reduction in the presence of 250 mg/l sodium chloride and 250 mg/l sodium sulfate. Nitrate reduction was retarded in the presence of chloride ion where as sulfate had little effect. In the presence of chloride ion the efficiency was reduced to 43% whereas with sulfate it was 63% and in simulated ground water the removal was above 71%. Hindrance of nitrate reduction in the presence of chloride ion may be due to production of oxidizing hypochlorite ion (Rajeswar and Ibanez, 1997) that forms at anode (Eqs. (2)–(4)). The little hindrance of sulfate might be due to competitive reduction of sulfate to H2S and other reduction products.

\[ 2\text{Cl}^- \rightarrow \text{Cl}_2 + 2e^- \]  
\[ \text{Cl}_2 + \text{H}_2\text{O} \rightarrow \text{HOCl} + \text{H}^+ + \text{Cl}^- \]  
\[ \text{HOCl} \rightarrow \text{OCl}^- + \text{H}^+ \]  

Ti is a nontoxic metal (USEPA, 1998) and thus is suitable for drinking water treatment. Anodic corrosion rates of Ti at pH 7 and 9 were found to be 8.48 and 8.55 moles Ti/mole of NO3-N reduced, respectively.

Table 2

Nitrate removal in the presence of anions

<table>
<thead>
<tr>
<th>Time (h)</th>
<th>Percentage NO3-N removed</th>
<th>SGW + 250 mg/l NaCl</th>
<th>SGW + 250 mg/l Na2SO4</th>
<th>SGW</th>
</tr>
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<tr>
<td>1</td>
<td>5.35</td>
<td>11.71</td>
<td>17.37</td>
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<tr>
<td>2</td>
<td>16.96</td>
<td>20.72</td>
<td>37.81</td>
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</tr>
<tr>
<td>3</td>
<td>26.64</td>
<td>37.84</td>
<td>53.81</td>
<td></td>
</tr>
<tr>
<td>4</td>
<td>36.82</td>
<td>51.35</td>
<td>62.81</td>
<td></td>
</tr>
<tr>
<td>5</td>
<td>42.85</td>
<td>63.06</td>
<td>71.27</td>
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</tr>
</tbody>
</table>
3.5. Redox potential

Oxidation reduction potential (ORP) was monitored during electrolysis with aluminum, iron and titanium electrodes at pH 7 and has been shown in Fig. 5. The cell with iron electrode exhibited the maximum negative potential (ORP = −463 mV) followed by aluminum (ORP = −245 mV) and titanium (ORP = −206 mV). It is evident from standard electrode potential values (Chebotareva and Nyokong, 1997) that formation of ammonia occurs at negative potential whereas the formation of nitrogen gas is feasible at positive potential. It seems that under high reducing environment (ORP = −463 mV) with iron electrodes nitrate rapidly gets converted to ammonia without buildup of intermediate products, as evidenced by negligible concentration of nitrite in aqueous phase. It may also be noted that the rate at which nitrogen (an intermediate product during denitrification) would strip/escape to the atmosphere is a function of the degree of supersaturation of nitrogen attained in the aqueous phase, and if the rate of nitrogen transformation to ammonia is rapid, then possibly supersaturation of nitrogen might not occur in the aqueous phase. Then in such a case all the nitrate-N present in aqueous phase would be converted to ammonia, which is observed in Fig. 2(a). Transformation of nitrate to products other than ammonia with titanium is significant and is explainable with higher ORP values (−206 mV), i.e., the least reducing condition that prevails inside the reactor when compared to aluminum and iron systems. At low reducing conditions the rate of conversion of nitrite is also slow and this is obvious from accumulation of nitrite in aqueous phase (Fig. 4) and may be that nitrogen supersaturation may have occurred leading to escape of nitrogen into the atmosphere. The conjecture is supported by the reduction of total-N observed in Fig. 4.

Similarly with aluminum electrodes where the reducing condition is intermediate between iron and titanium, the reduction of nitrite to ammonia is slower compared iron which is evident from accumulation of nitrite (Fig. 3).

3.6. Kinetics studies

Kinetics studies of any process has significant role in determining the hydraulic retention time in any reactor system to achieve desired removal. The disappearance of nitrate at low concentrations and low current density may follow first-order kinetic model where the rate is proportional to the dissolved contaminant concentration [C].

\[
d\frac{C}{dt} = -k_{obs}[C],
\]

where \( k_{obs} \) is the observed first-order rate coefficient. Integration of equation results in

\[
[C] = [C_0]e^{-k_{obs}t},
\]

where \([C_0]\) is the initial concentration of dissolved contaminant. Observed first-order rate coefficients have been calculated from linear regression of ln ([C]/[C_0]) vs. time and charge with \([C_0]\) set equal to the measured concentration at \( t = 0 \). The values of rate constants have been given in Table 3. Electrochemical nitrate removal is similar to nitrate removal with zero valent iron which has been described as a method for denitrification of aquifers and groundwater (Delvin et al., 2000; Huang et al., 1998; Schlicker et al., 2000; Siantar et al., 1996). Both these nitrate reduction processes are based on electron donation principle. Although the kinetics for nitrate removal with zero-valent iron has been described as first order the rate constant with metallic iron at pH 7.5 \((1.3 \pm 0.24) \times 10^{-1} \text{h}^{-1}\) is very less compared to electrochemical process shown in Table 3.

The presence of soluble iron and very high metallic iron surface area is necessary for denitrification process to be effective with metallic iron and sometimes an initial lag period has been observed at higher pH (Alowitz and Scherer, 2002). The observed electrochemical nitrate removal with iron electrode is not in agreement with reports of nitrate reduction by metallic iron where the reduction nearly ceases at pH > 5 and maintenance of pH with organic buffer has been reported (Zaeideh and Zhang, 1998). The higher removal at low pH with zero valent iron is related to its corrosion because at high pH surface passivation of iron particles occurs and iron corrosion ceases above pH 8.8 (Cheng et al., 1997). It has also been reported that rate constant tends to zero at pH 9 with metallic iron (Alowitz and Scherer, 2002). But the same effect was not observed during electrochemical process because electron transfer does not get affected by pH due to the applied electric potential.
It can be observed from Table 3 that the rate constants increase with increase in current density. Whereas, the rate constant when expressed with respect to charge (Coulombs) passed per liter are nearly same (Table 3). Thus charge density (current passed per unit volume) seems to be an appropriate parameter and can be considered as a design criterion. This is at variance with the earlier reports where current density has been considered as an important design parameter (Gender et al., 1996; Li et al., 1988a, b). Approximately 1 Coulomb of charge has to be passed/0.005 mg nitrate-N removal irrespective of electrode materials and experimental conditions.

3.7. Economic analysis

Preliminary estimate of the cost of denitrifying groundwater by electrochemical process has been done considering the energy cost and the cost of electrode. From the kinetic constants obtained in the study, amount of electric power required for desired nitrate reduction can be computed. The cost of electrical energy is variable in different parts of the world. Considering the cost of electrical energy as 8 cents (US $0.08) per KWH the associated energy cost for denitrifying 10 l of water by the process at neutral pH is found to be $0.075 (for reducing nitrate concentration from 100 mg/l nitrate-N to 10 mg/l nitrate-N); $0.40 (for reducing nitrate concentration from 30 mg/l nitrate-N to 10 mg/l nitrate-N). Cost of electrode materials for the process can be assessed from electrode corrosion rate. In the year 2002 the average cost of iron sheet was US $0.70 per kg, the average cost of aluminum sheet was US $1.6 per kilogram, and the average cost of titanium sheet was $30 per kg. Considering these prices the cost of Ti corroded during the process of removing nitrate level from 100 mg/l nitrate-N to 10 mg/l nitrate-N from 10 l of water is $0.75, and the costs of Al and Fe for similar removal are $0.04 and $0.025, respectively. Though the total operating cost of electrochemical process might be slightly higher but further improvements of the process might lead to reduction in cost.

4. Conclusions

This study demonstrated the applicability of electrochemical process for denitrification of ground water. The following conclusions are drawn based on the present study.

- Iron and aluminum mostly transform nitrate to ammonia where as titanium transforms nitrate to nitrogen.
- Strong reducing conditions (low ORP) favors formation of ammonia where as nitrogen and other oxide

### Table 3

Kinetics of electrochemical nitrate removal

| Sl. no. | Electrode | NO$_3$ Con. (mg/l) | Current density (mA/cm$^2$) | pH | $k_{obs}$ (Hour$^{-1}$) | $R^2$
<table>
<thead>
<tr>
<th></th>
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<tbody>
<tr>
<td>1</td>
<td>Fe</td>
<td>100</td>
<td>50</td>
<td>7</td>
<td>0.2896</td>
<td>0.9536</td>
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<tr>
<td>2</td>
<td>Fe</td>
<td>98</td>
<td>50</td>
<td>7</td>
<td>0.3062</td>
<td>0.9812</td>
</tr>
<tr>
<td>3</td>
<td>Fe</td>
<td>200</td>
<td>10</td>
<td>7</td>
<td>0.0389</td>
<td>0.9517</td>
</tr>
<tr>
<td>4</td>
<td>Fe</td>
<td>205$^a$</td>
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<td>7–12</td>
<td>0.4168</td>
<td>0.9962</td>
</tr>
<tr>
<td>5</td>
<td>Fe$^b$</td>
<td>197</td>
<td>140</td>
<td>7–11</td>
<td>0.1798</td>
<td>0.9813</td>
</tr>
<tr>
<td>6</td>
<td>Fe$^b$</td>
<td>185$^a$</td>
<td>140</td>
<td>8–10</td>
<td>0.3695</td>
<td>0.9799</td>
</tr>
<tr>
<td>7</td>
<td>Fe$^b$</td>
<td>185$^a$</td>
<td>140</td>
<td>4</td>
<td>0.1788</td>
<td>0.9941</td>
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<tr>
<td>8</td>
<td>Al</td>
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<td>0.2238</td>
<td>0.9956</td>
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<tr>
<td>9</td>
<td>Al</td>
<td>99.5</td>
<td>50</td>
<td>7</td>
<td>0.3024</td>
<td>0.9783</td>
</tr>
<tr>
<td>10</td>
<td>Al</td>
<td>104</td>
<td>50</td>
<td>9</td>
<td>0.3634</td>
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</tr>
<tr>
<td>11</td>
<td>Al$^b$</td>
<td>191$^a$</td>
<td>140</td>
<td>8–10</td>
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<tr>
<td>12</td>
<td>Al$^b$</td>
<td>191$^a$</td>
<td>140</td>
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<tr>
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<td>Al$^b$</td>
<td>105</td>
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<td>0.1881</td>
<td>0.9952</td>
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<tr>
<td>14</td>
<td>Ti</td>
<td>99</td>
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<td>0.2482</td>
<td>0.996</td>
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<tr>
<td>15</td>
<td>Ti</td>
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<td>9</td>
<td>0.3609</td>
<td>0.9331</td>
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<tr>
<td>16</td>
<td>Ti$^c$</td>
<td>93</td>
<td>50</td>
<td>7</td>
<td>0.1988</td>
<td>0.9878</td>
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<tr>
<td>17</td>
<td>Ti$^d$</td>
<td>111</td>
<td>50</td>
<td>7</td>
<td>0.1297</td>
<td>0.9982</td>
</tr>
</tbody>
</table>

$^a$0.12 M NaHCO$_3$ has been added to SGW.
$^b$5.75 cm$^2$ electrode face and 300 ml cell.
$^c$With 250 mg Na$_2$SO$_4$.
$^d$With 250 mg NaCl.
formation occurs at moderate or low reducing conditions (high ORP).

- The transformation of nitrate to nitrogen is largely dependent on electrode material used and the rate of nitrate transformation is insignificantly affected by pH in the pH range 7–9.
- The electrochemical nitrate removal at nitrate concentration studied follows first order rate law. From kinetics studies it is clear that charge density (current passed per unit volume) is an appropriate design parameter.

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References


