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Electrokinetic Soil Remediation: Challenges and Opportunities

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Abstract: Electrokinetic remediation is effective for extraction of contaminants, such as heavy metals, from fine grained deposits. However, several challenges have led to limited field implementation in the US over the past two decades. The reasons include lower risk of exposure to adsorbed contaminants through groundwater, cost of treatment, technical challenges in difficult soil, and complex geochemical conditions, and the need for acidification to induce desorption. To address practical implementation, a Reactive Transference Factor can be used to identify conditions that are favorable for electrokinetic remediation. The factor measures the reactive transport rates relative to the electrical conductivity of the soil. Other opportunities are identified for implementation of electric-based methods for remediation beyond heavy metal extraction. These include enhancement of bioremediation, which is favorable for remediation of hot spots or source areas, and the development of reactive electrochemical barriers, particularly for the treatment of contaminated groundwater plumes.

Keywords: Electrokinetic remediation, groundwater, soil, bioremediation, electroosmosis

INTRODUCTION

Soil and groundwater contamination from various sources, both man made (e.g. manufacturing, agriculture, and mining byproducts) and naturally occurring (e.g. arsenic), continues to be a significant and costly

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Table 1. Estimated number of sites and remediation cost in the US ['04-'33] (Pachon, et al., 2008)

Program	No. of sites	Cleanup cost
NPL	1,146–1926	$41–103 B
RCRA, CA	3,829	$31–58 B
RCRA, UST	215,826–395,827	$27–49 B
DOD	6,199	$31 B
DOE	5,000	$73 B
Civilian Agencies	3,000	$15–22 B
State and Private	150,000	$30 B
Total Range Middle Value	385,001–565,781 475,000	$248–366 B $322 B

problem for both the developed and the developing world. In the US, several programs and activities have resulted in more than 500,000 contaminated sites (Table 1) with an estimated cleanup cost that may exceed $300 billion over the next 25 years (1). The type of contaminants that are present at these sites include several groups of chemicals, such as volatile organic compounds (VOCs), polycyclic aromatic hydrocarbons (PAHs), metals, polychlorinated biphenyls (PCB's), pesticides, radionuclides. Although significant efforts and resources have been invested in developing and implementing soil and groundwater cleanup technologies, several technical and practical challenges have limited the success of these technologies. For example, since 1983, the Environmental Protection Agency (EPA) has listed 1,579 sites on its National Priorities List (NPL) of Superfund sites, yet only 321 have been cleaned up and removed from the list (2). Cleanup efforts are increasingly more difficult because of lack of funding due to recent economic woes. Since 2000, an average of 42 NPL sites per year were moved into the construction phase as compared to an average of 79 sites per year in the 1990's (2). In 2007, only 24 NPL sites were moved into the construction phase.

Remediation of contaminated fine grained deposits and heterogeneous, leaky aquifers is particularly difficult. After more than three decades of laboratory research and field studies/trials, such deposits still pose a major challenge for environmental restoration of contaminated land. The problem is two-fold. First, it is impractical to deliver or transport aqueous chemicals through or out of soil layers with low hydraulic conductivity. Second, these layers continue to serve as source zones or hot spots because of contaminant sorption, such as heavy metals, chlorinated solvents and PAHs, in fine-grained soils or because of the contaminants' presence as nonaqueous phase liquids (NAPLs) within the aquifers. Because of the difficulties in treating these source zones (due to sorption or NAPLs), remediation activities in the US have focused on the containment of the source zones coupled with active or passive treatment, such as permeable reactive barriers, bioremediation, or enhancement of natural attenuation, for aqueous and mobile contaminants. Mobile contaminants that can be treated by such technologies include chlorinated ethenes, chromium, arsenic, uranium, perchlorate. Some technologies, such as electrokinetics and phytoremediation, have targeted the hot spots, where contamination exists in low hydraulic conductivity zones. Although they have been proven successful, field implementation of electrokinetic remediation has lacked behind other technologies that target mobile contaminants. The primary reasons are

a. the high and immediate threat of aqueous contamination plumes on groundwater resources, and
b. the technical and practical difficulties in addressing the source zone contamination.

ELECTROKINETIC REMEDIATION DEVELOPMENT

Electrokinetic transport in soil was considered in the late 1980s as a mechanism to facilitate and accelerate the extraction of contaminants from contaminated soils with relatively low hydraulic conductivity (e.g., fine-grained soils). In mid to late 1980s and early 1990s, laboratory studies evaluated the role of electroosmosis and ion migration for extraction of heavy metals and ions (3–11). Over the past 10 years, several laboratory studies have emphasized the need to enhance the process in order to improve extraction (12–17). Other laboratory studies (e.g., 18, 19) have evaluated the complex effect of site-contaminated soil, shifting their focus from laboratory synthetic samples to site-contaminated soils. While the success of the technology is sensitive to many physicochemical variables, such as cation exchange capacity, surface charge and geochemistry, its major advantages include:

a. it can be implemented *in situ* with minimal soil disruption,
b. it is well suited for fine-grained, heterogeneous media, where other techniques are ineffective, and
c. the technology minimizes the post-treatment volume of waste material.

A few pilot-scale and field demonstration/validation studies have been performed, mostly in the 1990s. These include removal of chromium (VI) from unsaturated soil (moisture content in the range of 2 to 12% by weight) beneath Chemical Waste Landfill (CWL) (20); extraction of chromium from a site at Point Mugu, California (21); extraction of lead from soils contaminated at a firing range (22); and extraction of uranium from a site at Oak Ridge K-25 Facility, Oak Ridge, Tennessee (23). Difficulties and challenges were reported in mercury extraction, but good transport of lead and chrome was reported in a field demonstration at Old TNX Basin, Savannah River Site, South Carolina (23). Successful demonstration was reported by Geokinetics International, Inc. (11) for *in situ* and *ex situ* electrokinetic extraction of metals and organics. Field studies have also been conducted for extraction of organics for contaminated clay. Electroosmotic extraction using layered horizontal electrodes or the Lasagna^{TM} process was implemented to treat an area of 14 m^2 up to a depth of 5 m at the Paducah Gaseous Diffusion Plant (PGDP), Paducah, Kentucky. The process reduced trichloroethylene (TCE) concentration in clay from the original range of 100 to 500 parts per million (ppm) to an average concentration of 1 ppm (24).

Implementation of electrokinetic remediation is site-specific; thus the design, implementation and performance vary from site to site. Several variables will be required to assess the feasibility, practicality, and cost efficiency for each site. The variables include the contribution of ion migration, electroosmosis, geochemistry, and boundary conditions (electrolysis) on the reactive transport rates. These variables are discussed below in addition to the impact of transient nonlinear conditions between the electrodes.

Reactive Transport: Mass Transference Factor

A major advantage of electrokinetic remediation is that transport by electroosmosis and ion migration is significant in fine-grained soils. Another advantage is the possibility of developing a relatively uniform transport in heterogeneous deposits. Although ion migration rates are higher in sand than in clays, the differences are not as significant when compared with transport by hydraulic gradient, where the hydraulic conductivity of sandy soils is orders of magnitude higher than clayey soils. Thus hydraulic gradients are ineffective in removing contaminants from heterogeneous soils because of the limited transport in clays. Figure 1 shows comparisons between transport under hydraulic and electrical gradients in sands and clay. A relatively uniform transport can be produced in heterogeneous soils by ionic migration. However, transport in soils is more difficult to assess when accounting for the reactivity of contaminants.

A reactive mass transference factor can be developed for specific contaminants in soils in order to evaluate the performance and develop

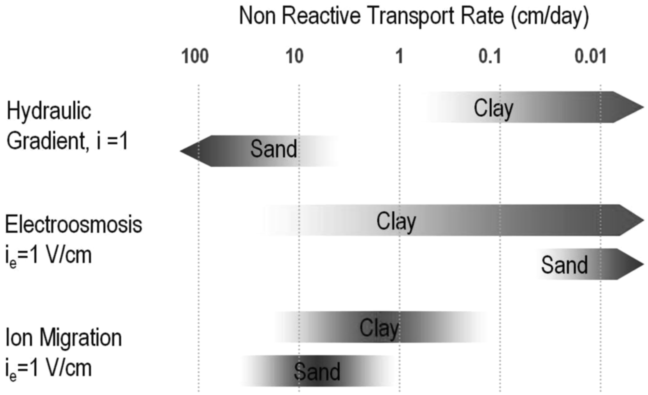


Figure 1. Comparison of transport rates by hydraulic and electrical gradients in sand and clay.

design strategies for electrokinetic remediation. The transference factor is evaluated by measuring the rate the mass flux of a specific chemical species relative to the total charge flow rate (or electric current). Assuming diffusion is negligible compared to ion migration and electroosmosis, the fluxes are calculated by:

$$J_i = (u_i^* + k_e)c_i i_e$$

$$I = \sigma^* i_e$$

J_i: Total mass flux of species i(mg/day/m^2)

c_i : Concentration of species i(Molar)

u_i^*: Effective ionic mobility of i(m^2/day/Volt)

I: Electric current density (Amp/m^2)

k_e: Coefficient of electroosmotic permeability (m^2/day/Volt)

σ^* : Effective electrical conductivity (Siemens/m)

i_e : Electrical gradient (V/m)

The equivalent mass transference factor, representing the rate of mass flux per charge, in soil is given by:

$$\text{Mass transference factor } (g/C) = \frac{J_i}{I} = \frac{u_i^* + k_e}{\sigma^*} c_i$$

This factor has units of gram per Coulomb; however, multiplying by the valence and Faraday's constant will result in a non-dimensional parameter that is equivalent to transference number defined in electrochemistry literature (in soils, the number accounts for transport due to both ion migration as well as electroosmosis).

Ions transport in soil is different than in electrolytes due to the reactivity of ions on soil surface and complex geochemistry. Reactions will vary and may include sorption, biological transformation, chemical redox transformation, and/or precipitation/dissolution. The reactivity rates and kinetics will also vary. However, to simplify, we will only introduce a delay factor R that accounts for the delayed or limited transport due to reactivity. The reactive mass transference factor is:

$$\text{Reactive Transference Factor} = \frac{u_i^* + k_e}{R\sigma^*} c_i$$

R accounts for delay in transport due sorption, decay, precipitation/dissolution, etc.

Consider, for example, saturated silty clay with the following properties: porosity is 0.4, tortuosity is 0.3, electrical conductivity is 0.02 S/m, and the coefficient of electroosmotic conductivity is $1 \times 10^{-9}\,\mathrm{m}^2/\mathrm{V\,sec}$. Assume the contaminant ionic mobility (at infinite dilution) to be $5 \times 10^{-8}\,\mathrm{m}^2/\mathrm{V\,sec}$ and a reactivity "delay" factor (R) that is equal to 2. If the pore fluid concentration of the contaminant is 100 mg/L, then the reactive transport factor is calculated as 0.0175 mg/C.

High reactive transference factors indicate effective transport and favorable conditions for electrokinetic remediation. This factor will vary by orders of magnitude for different conditions. The factor provides an insight into the effects of the variable (primarily electroosmosis, ionic migration, concentration, electrical conductivity and reactivity) on the performance of electrokinetic remediation. These variables are transient, heterogeneous, and interdependent. They are transient because they vary over time due to changes in geochemistry. They are heterogeneous because they will develop a non-uniform distribution between the electrodes. They are interdependent because changes in one variable (e.g. electrical conductivity) will induce changes in other variables (e.g. electroosmotic conductivity). A discussion of the variables effect on the process is provided below.

Nonlinear Behavior

Electrolysis will produce extreme pH and redox conditions at the electrodes, and thus defines the boundary conditions for the process. The anode

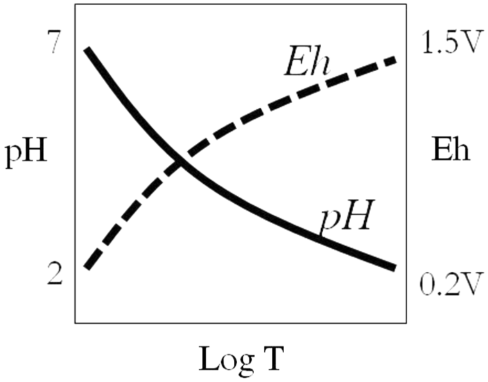


Figure 2. Changes in pH and EH at the anode (Boundary conditions).

will develop an acidic pH and an oxidizing redox potential (Fig. 2) while the cathode will develop an alkaline pH and a reducing redox potential (Eh). The extreme boundary conditions will cause transient and nonlinear distributions in the geochemistry profile, particularly pH, electrical conductivity, and redox potential, across the electrodes. This nonlinearity will result in a transient and non-uniform electric potential distribution and complex electro-chemico-mechanical behavior. It is important to emphasize these changes and the need to better understand these complex relationships.

Consider for example the changes in electroosmotic conductivity, as discussed by Alshawabkeh et al., 2004. Usually, the electroosmotic conductivity is calculated by assuming that all electric potential is effective in inducing electroosmosis, a linear distribution of the electric potential between the electrodes; and that steady state electric field conditions are maintained. These assumptions are very simplifying and do not represent actual conditions.

When the process is used for extraction of heavy metals, the high pH developed at the catholyte due to electrolysis causes precipitation. Organic acids have been proposed and used to prevent the increase in pH and hydroxide precipitation. However, if no pH control is used, the high pH and hydroxide precipitation decrease the ionic electric conductivity of the soil, causing most of the voltage drop and electroosmotic flow that occur near the cathode. The low ionic strength of pore fluid in sections near the cathode causes a thicker double layer and increased electroosmotic flow rate. On the other hand, acidification of the sections near the anode causes a decrease in the negative surface charge, zeta potential and electroosmotic flow. Consequently, high electroosmosis

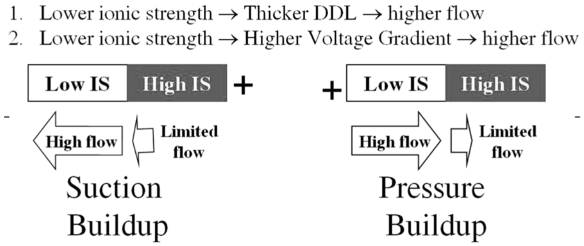


Figure 3. Impacts of formation of non-uniform conditions on electroosmosis and porewater pressure.

occurs in sections near the cathode compared to sections near the anode. Since the hydraulic conductivity of the soil is very low, the sections near the anode act as an impermeable medium, preventing the necessary flow, and producing a negative pore pressure in the soil. The development of negative pore pressure during electrokinetic extraction of heavy metals was reported by (25,26).

Other applications, such as additives injection for enhanced bioremediation, produce different conditions. For example, injection of lactic acid at the cathode causes buildup of a nonlinear electroosmotic flow and development of positive pore water pressure (27). The electroosmotic water quantities collected during a 24-hour period show that higher levels of electroosmosis occur at the anode side than the cathode side. Measurements of the electric conductivity and the k_e profile across the cell indicate that k_e decreases with increasing electric conductivity. The value of k_e is six times higher in sections near the anode ($k_e \sim 6 \times 10^{-5}\,cm^2/V \cdot s$) s) compared to sections close to the cathode ($k_e \sim 1 \times 10^{-5}\,cm^2/V \cdot s$). Figure 3 shows two potential cases where the nonlinearity in the physicochemical condition can induce either positive or a negative pore water pressure.

In summary, the development of transient non-uniform geochemical profiles between the electrodes is critical. It will produce transient changes on transport, reactivity, and volume change. Only a few studies have attempted to understand and predict these changes. There is a need to develop an improved fundamental understanding of these changes for specific applications.

Challenges

Electrokinetic remediation is very effective for remediation of difficult soils and source areas that are contaminated with bound contaminants, such as heavy metals. However, there are several reasons for its limited field implementation in the US over the past two decades. These reasons are primarily associated with the application of the heavy metal extraction process.

There has been limited interest in extraction of adsorbed contaminants, such as some heavy metals, from soils. The main reasons are the technical challenges and high costs that are required for cleanup of the heavy metal-contaminated source areas. Extraction requires desorption or dissolution followed by heavy metal transport. This is feasible if the concentrations are relatively high, i.e., beyond the sorption capacity of the soil. Once the concentrations decrease below the sorption capacity, then higher energy will be required in addition to substantial changes in the soil geochemistry (e.g., acidification or injection of additives, chelating agents, etc.). That will require higher costs (energy and chemicals) and longer treatment times, limiting the advantages of electrokinetic remediation. Unless there is direct contact with soils with adsorbed contaminants, there is limited exposure and risk. Because of the technical challenges, high costs for remediation and limited exposure/risks, the short- or long-term remediation actions at these sites are not expected to be priorities.

When extracting adsorbed heavy metals, electrokinetic remediation promotes the transport and use of acids generated at the anodes for desorption. Soil acidification and acid injection are not favorable for remediation. The acids will significantly alter the soil geochemistry as well as the soil microstructure.

Another issue for heavy metal extraction is the restriction on the radius of influence, which will lead to relatively close electrode spacing. Transport of heavy metals under electric currents in fine-grained soils is orders of magnitude higher than under hydraulic gradients. However, this transport is still on the order of a few centimeters per day at best, primarily because of the effect of adsorption leading to high retardation factors. That will lead to smaller electrode spacing, a high number of wells and higher cost.

The process of heavy metal extraction by electrokinetic remediation is complex. Transport processes are well-defined, but complexity arises from the nonlinear and transient geochemistry between electrodes during processing. As the pH front moves under electric currents, nonlinear reactive transport will develop. Acidification will change the distribution of ionic strength, electrical conductivity, voltage, zeta potential, and electroosmosis. It becomes difficult to treat the soil between the electrodes as a uniform continuum. Analysis and design should account for the transient and nonlinear changes in the system. This will lead to an efficient and a successful implementation of the technology.

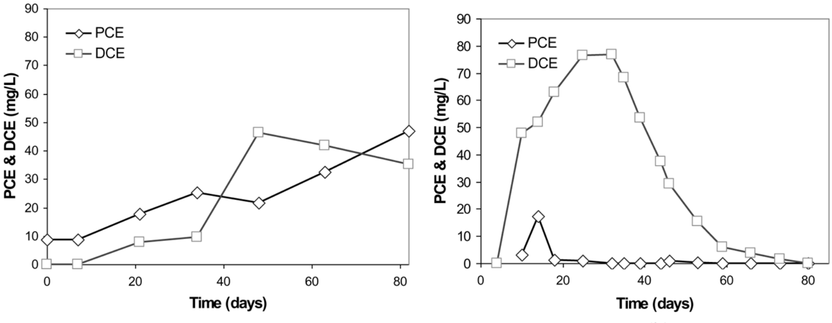
Opportunities

Electrokinetic and electrochemical processes are very powerful for inducing controlled changes in soil and groundwater. However, most of the effort over the past two decades has focused on extraction of contaminants from soils by electroosmosis and ionic migration. It is necessary to explore the potential of these powerful processes beyond contaminant extraction. Here, two potential applications are discussed. The first targets source zones by stimulation of bioremediation. The second targets mobile contamination plumes by development of reactive electrochemical barriers.

EK Enhanced Bioremediation

Aerobic or anaerobic bioremediation can be engineered for different types of contaminants in soil and groundwater. Contaminant type and groundwater biogeochemistry typically determine the best conditions for bioremediation. For example, aerobic conditions are better suited for biodegradation of polycyclic aromatic hydrocarbons (PAH's) in groundwater, while anaerobic reducing conditions are more suited for biodegradation of chlorinated ethenes. In either case (aerobic or anaerobic), bioremediation requires injection of additives, such as electron acceptors or donors, to stimulate biological activity. A key limitation for bioremediation is effective injection and mixing of these additives or biological amendments, especially for low permeable soils and heterogeneous subsurface (28). The ineffectiveness of typical delivery methods (e.g., hydraulic injection) is due to low hydraulic conductivity and the degradability of the organic additives. In some cases, the additives, e.g., lactate (29), are not detected in soil under hydraulic injection because their degradation rates are much higher than the transport rate. Electrokinetics can produce uniform transport and mixing of additives for bioremediation in heterogeneous and fine grained deposits.

The concept of ionic injection by electric fields for enhanced bioremediation was introduced in the mid-1990s. High additive transport rates were measured in soils, such as sulfate transport of up to 20 cm/d across 80 cm beds of fine sand (30), acetate and nitrate across 40 cm low permeable silty loam and clay samples (hydraulic conductivity less than 3 cm/year) at rates of 2.0–2.6 cm/day (31), and citrate migration through contaminated marine sediments over a distance of 4.6 meters at a rate of 30 cm/day (32). Reactive (accounting for delay due to sorption and transformation) transport rate of lactate on the order of 5 cm/day in sand and 3.7 cm/day in clay are measured (29) under electric currents.



(a) (b)

Figure 4. PCE and DCE concentration in clay samples due to bioremediation by lactate injection using (a) no electricity (control) and (b) electrokinetic injection.

Nutrient amendment by electrokinetics has been reported (17) for the bioremediation of chromium-contaminated soil. The results demonstrate that transport under electric fields is on the order of a few centimeters per day for reactive, degradable anions and on the order of tens of centimeters per day for nonreactive tracers.

A recent study was conducted to evaluate the enhancement of *in situ* bioremediation of PCE with electrochemical injection of lactate, a commonly used electron donor (33) for anaerobic biodegradation. The soil was low plasticity (CL) clay with 1.4% organic content. A 40 cm long soil sample, contaminated with perchloroethylene (PCE), was prepared. Lactate was introduced at the electrodes while mixing the electrolytes to maintain neutral pH conditions at the boundaries. The results showed that in 60 days, lactate transport under electric current induced continuous buildup of DCE as a result of PCE transformation (Fig. 4). For the control experiment (no electricity), PCE transformation was limited and after 60 days, PCE was present in the soil with limited transformation and formation of DCE. In this study, the Reactive Transference Factor for lactate was calculated to be on the order of 0.052 mg/C. This factors is calculated based on reactive transport rate ($u_i^* + k_e/R$) of 3.7 cm2/V day, target concentration of 1000 mg/L, and average electrical conductivity of 825 μS/cm (0.0825 S/m).

Electrochemical Redox Barriers

Electrochemical redox reactions can be engineered to develop reactive permeable redox barriers for *in situ* remediation of contaminated groundwater. Electrolysis under DC currents causes chemical reduction

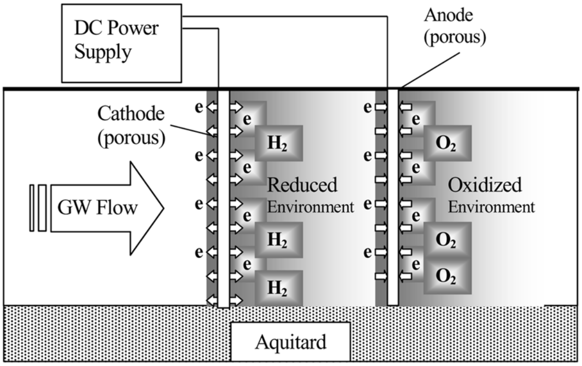


Figure 5. A schematic of permeable electrochemical redox barriers.

and production of H_2 at the cathode and chemical oxidation and production of O_2 at the anode. The process will depend on the groundwater chemistry and the efficacy of electrolysis redox and gas production (e.g., hydrogen as an electron donor or oxygen as an electron acceptor) under direct currents. The process will raise several interesting scientific questions, such as the impact of the electrochemical redox reactions on the abiotic transformation versus stimulation of biodegradation by the indigenous microbial populations. The process is particularly relevant for remediation of chlorinated ethenes in groundwater. In addition to abiotic transformation, it is possible to utilize two treatment zones for biological treatment: first anaerobic near the cathode as a result of hydrogen gas formation, followed by aerobic near the anode as a result of oxygen formation (Fig. 5).

For chlorinated ethenes, electrolysis reactions may cause abiotic transformation in groundwater. In the literature on electrolytic reduction of water/groundwater, the transformation of chlorinated ethenes is evaluated at the cathode, in circulated anolyte/catholyte or electrolytic flow systems using inert anodes (e.g., noble metals, and carbon) and the redox is dependent on electrode types and electrolyte conditions. Electrolytic reduction and dechlorination of contaminated groundwater at the cathode has been demonstrated using silver (34), iron (35,36), copper (36), palladium (37), Pt- or Pd-coated ceramic (38), bare and polymer-coated nickel (39) and mixed metal oxide (MMO)-coated titanium mesh cathodes (40). Several reductive pathways are possible; however, the primary pathway for TCE reduction by cathodic electrolysis on iron and palladized-iron may be indirect and involves atomic hydrogen as the reducing agent with TCE being reduced primarily to ethane and ethene (41).

The impact of electrolysis redox reactions on chlorinated solvents was observed in a field demonstration of electrokinetic extraction of metals at Naval Air Weapon Station, Point Mugu, California. The study was sponsored by the Department of Defense Environmental Certification and Technology Compliance Program (ESTCP). Two lagoons contaminated with chromium (up to 14,000 mg/kg), and cadmium (up to 1,900 mg/kg) were treated using 1 V/cm DC field. While the focus of this project was to demonstrate metal extraction, results also indicated that DC electrolysis reactions enhanced the reductive dehalogenation of chlorinated solvents (it was not clear if this was due to abiotic or biotic transformations). The disappearance of TCE, dibromomethane, and methylene chloride at cathode wells suggests that the reducing environment at the cathode promoted their dechlorination in less than 2 months. Furthermore, an accumulation of vinyl chloride (VC), a daughter product of TCE and dichloroethylene (DCE) reductive dechlorination, was also observed at locations away from the cathodes, suggesting the transformation of the parent compounds. Although the project was not designed to confirm the TCE transformation mechanism, the observations clearly demonstrate that a bioelectrochemical treatment zone could be engineered to remediate groundwater contaminated with chlorinated solvents.

An electrochemical redox barrier can provide significant advantages over other types of barriers because:

a. it provides a continuous source/sink of electrons,
b. the rates of redox reactions can be engineered by controlling electric current density,
c. the system can provide reducing as well as oxidizing (or sequential) conditions,
d. the system does not require injection of any reactive materials,
e. the electrode material is not consumed and may be reused, and
f. the effect on groundwater chemistry can be reversed at anytime by reversing the polarity of the electrodes.

The electric current required for treatment of contaminated groundwater can be calculated based on contaminant flow rate. Assuming groundwater conditions with initial contaminant concentration of c (ML^{-3}), the contaminant mass flow rate is:

$$J = cq \qquad (3)$$

Where J ($MT^{-1}L^{-2}$) is the contaminant mass flow rate per unit cross-sectional area of the aquifer, and q ($L^3T^{-1}L^{-2}$) is the groundwater flow rate per unit area. Using Faraday's Law for equivalence of mass and charge, the electric current required for redox reactions is:

$$I = \alpha FJ = \alpha Fcq \tag{4}$$

where I (Amp L^{-2}) is the electric current density and F (96,485 C/ mole e). The factor α (mole e/mole contaminant) account for electrolysis efficiency and biochemical redox requirements.

SUMMARY

Electrokinetic remediation is very effective for remediation of fine-grained deposits, heterogeneous soils, and source areas that contain strongly bound contaminants, such as heavy metals. However, reasons for limited field implementation in the US include lower risk of exposure to adsorbed contaminants through groundwater, higher costs of treatment, difficult soil and geochemical conditions, and the need for acidification to induce desorption.

A reactive mass transference factor can be developed for specific contaminants in soils to evaluate the performance and develop design strategies for electrokinetic remediation. The transference factor measures the rate of reactive mass transport by electroosmosis and ion migration relative to the electrical conductivity of the soil. High reactive transference factors indicate effective transport and favorable conditions for electrokinetic remediation. The factor provides insight into the effects of the variable (primarily electroosmosis, ionic migration, concentration, electrical conductivity and reactivity) on the performance of electrokinetic remediation. These variables are transient, heterogeneous, and interdependent.

Different boundary conditions are specified at the electrodes for different applications. The boundary conditions result in the development of transient non-uniform geochemical profiles between the electrodes which themselves produce transient changes on transport, reactivity, and volume change. Only a few studies have attempted to understand and predict these changes. There is a need to develop fundamental studies to assess the development and understanding of these changes for specific applications.

Most of the efforts over the past two decades have focused on extraction of contaminants by electroosmosis and ionic migration. It is necessary to explore the potential of these powerful processes beyond contaminant extraction. Two potential applications include stimulation of bioremediation to treat source zones and the development of reactive electrochemical barriers for cleanup of mobile contamination plumes in groundwater.

REFERENCES

1. Pachon; Carlos; Howe, R.; Purucker, T. (2008) The Critical Role of Data Management. Presentation, ConSoil 2008, 10th Annual UFZ/TNO Conference on Soil-Water Systems, June, 2008; Milan: Italy.
2. Hogue, C. (2007) Superfund Slowdown: Lagging pace of cleanups blamed on technical challenges and lack of money. *Chemical and Engineering News*, *85* (46): 41–44.
3. Runnells, D.D.; Larson, J.L. (1986) A laboratory study of electromigration as possible field technique for the removal of contaminants from groundwater. *Groundwater Monitoring Review*, *6* (3): 85–91.
4. Acar, Y.B.; Gale, R.J.; Putnam, G.; Hamed, J.T. (1989) Electrochemical Processing of Soils: Its Potential Use in Environmental Geotechnology and Significance of pH Gradients. *2nd International Symposium on Environmental Geotechnology*, Shanghai, China, May 14–17, Envo Publishing: Bethlehem.
5. Lageman, R.; Wieberen, P.; Seffinga, G. (1989) Theory and Practice of Electro-Reclamation, NATA/CCMS Pilot Study. *Demonstration of Removal Action Technologies for Contaminated Land and Groundwater*; Copenhagen: Denmark, May 9, 18 p.
6. Hamed, J.; Acar, Y.B.; Gale, R.J. (1991) Pb(II) Removal from kaolinite using electrokinetics. *Journal of Geotechnical Engineering, ASCE*, *112* (2): 241–271.
7. Pamukcu, S.; Wittle, J. K. (1992) Electrokinetic removal of selected heavy metals from soil. *Environmental Progress*, *11* (3): 241–250.
8. Alshawabkeh, A.N.; Acar, Y.B. (1992) Removal of contaminants from soils by electrokinetics: A theoretical treatise. *Environmental Science and Health*, *A 27* (7): 1835–1861.
9. Eykholt, G.R. (1992) Driving and complicating features of the electrokinetic treatment of contaminated soils. PhD Diss. Department of Civil Engineering, Univ. of Texas at Austin, Tex.
10. Runnells, D.D.; Wahli, C. (1993) In situ electromigration as a method for removing sulfate, metals, and other contaminants from groundwater. *Ground Water Monitoring & Remediation*, *13* (1): 121–129.
11. Lageman, R. (1993) Electro-reclamation. *Journal of Environmental Science and Technology*, *61* (13): 2648–2650.
12. Hicks, R.E.; Tondorf, S. (1994) Electrorestoration of metal contaminated soils. *Envir. Sci. & Tech.*, *28* (12): 2203–2210.
13. Acar, Y.B.; Alshawabkeh, A. N. (1996) Electrokinetic Remediation: I. pilot-scale tests with lead spiked kaolinite. *ASCE, J of Geotechnical Engineering*, *122* (3): 173–185.
14. Wong, J.S.; Hicks, R E.; Probstein, R.F. (1997) EDTA- enhanced electroremediation of metal contaminated soils. *Journal of Hazardous Materials* Special Edition on Electrochemical Decontamination of Soil and Water, *55* (1): 61–79.
15. Yeung, A.; Hsu, C.; Menon, R. (1996) EDTA-enhanced electrokinetic extraction of lead. *ASCE, Journal of Geotechnical Engineering*, *122* (8): 666–673.

16. Reddy, K.R.; Xu, C.Y.; Chinthamreddy, S. (2001) Assessment of electrokinetic removal of heavy metals from soils by sequential extraction analysis. *Journal of Hazardous Materials*, *84* (2–3): 85–109.
17. Reddy, K.R.; Chinthamreddy, S. (2003) Sequentially enhanced electrokinetic remediation of heavy metals in low buffering clayey soils. *Journal of Geotechnical and Geoenvironmental Engineering, ASCE*, *129* (3): 263–277.
18. Alshawabkeh, A.N.; Puppala, S.; Acar, Y.B.; Gale, R.; Bricka, R.M. (1997) Effect of solubility on enhanced electrokinetic extraction of metals. *Proceedings of "In Situ Remediation of the Geoenvironment - In Situ Remediation '97,"* ASCE, Geotechnical Special Publication No. 71, 532–544.
19. Puppala, S.; Alshawabkeh, A.N.; Acar, Y.B.; Gale, R.; Bricka, R. (1997) Enhanced electrokinetic remediation of high sorption capacity soils. *Journal of Hazardous Materials*, *55*: 203–220.
20. Lindgren, E.R.; Hankins, M.G.; Mattson, E.D.; Duda, P.M. (1998) Electrokinetic Demonstration at the Unlined Chromic Acid Pit. Report Abstract. SAND97–2592. Sandia National Laboratory.
21. Gent, D.; Larson, S.; Alshawabkeh, A.N.; Bricka, M.; Granade, S. (2004) Bench and field-scale demonstration of chromium and copper remediation by electrokinetics. *Journal of Hazardous Materials*, *110*: 53–62.
22. Alshawabkeh, A.N.; Bricka, M.; Gent, D. (2005) Pilot-scale electrochemical cleanup of lead contaminated soils. *ASCE, Journal of Geotechnical and Geoenvironmental Engineering*, *103* (3): 283–291.
23. EPA. (1995) In Situ Remediation Technologies: Electrokinetics. Report No. EPA542-K-94-007, April 1995, Environmental Protection Agency, Office of Solid Waste and Emergency Response, Technology Innovation Office, Washington, D.C., 20 p.
24. Ho, S.V.; Athmer, C.; Sheridan, P.W.; Hughes, B.M.; Orth, R.; McKenzie, D.; Brodskey, P.H.; Shapiro, A.M.; Thornton, R.; Salvo, J.; Schultz, D.; Landis, R.; Griffith, R.; Shoemaker, S. (1999) The lasagna technology for in situ soil remediation. 2. large field test. *Environmental Science and Technology*, *33* (7): 1092–1099.
25. Eykholt, J.; Daniel, D.E. (1994) Impact of system chemistry on electroosmosis in contaminated soil. *Journal of Geotechnical Engineering. ASCE*, *120* (5): 797–815.
26. Alshawabkeh, A.N.; Acar, Y.B. (1996) Electrokinetic remediation: II. theoretical model. *ASCE, Journal of Geotechnical Eng.*, *122* (3): 186–196.
27. Alshawabkeh, A.N.; Sheahan, T.C.; Wu, X. (2004) The effects of DC field application on soft soil properties. *Journal of Mechanics of Materials, Special Issue on Chemo-Mechanical Coupling*, *36*: 53–465.
28. Sims, J.L.; Suflita, J.M.; Russell, H.H. (1992) In Situ Bioremediation of Contaminated Ground Water. Ground Water Issue Paper. EPA/540/S-92/003. R.S. Kerr Environmental Research Laboratory, Ada, OK. 11 p.
29. Wu X.; Alshawabkeh, A.N.; Gent, D.; Larson, S.; Davis, J. (2007) Rates of lactate transformation and transport in soil under DC fields. *ASCE Journal of Geotechnical and Geoenvironmental Engineering*, *133* (12): 1587–1596.

30. Acar, Y. B.; Rabbi, M. F.; Ozsu, E. E. (1997) Electrokinetic injection of ammonium and sulfate ions into sand and kaolinite beds. *J. Geotech. Geoenviron. Eng.*, *123* (3): 239–249.
31. Gent, David, B.; Bricka, R.; Mark, R.; Truax, D.D.; Zappi, Mark. (2001) Electrokinetic Movement of Biological Amendments Through Natural Soils to Enhance In-Situ Bioremediation. *In Situ and On-Site Bioremediation, Sixth International Symposium.* San Diego, CA, June 4–7, 2001.
32. Granade, Steve; Gent, David, B.; Larson, Steve (2002) Electrokinetic Remediation of Contaminated Sediments. First International Conference on Remediation of Contaminated Sediments, Venice: Italy, October 8, 2002.
33. Song, D.L.; Conrad, M.E.; Sorenson, K.S.; Alvarez-Cohen, L. (20002) Stable carbon isotope fractionation during enhanced in situ bioremediation of trichloroethene. *Environ. Sci. Technol.*, *36*: 2262–2268.
34. Criddle, C.S.; McCarty, P.L. (1991) Electrolytic model system for reductive dehalogenation in aqueous environment. *Environ. Sci. Technol.*, *25*: 973.
35. Scherer, M.M.; Westall, J.C.; Ziomek-Moroz, M.; Tratnyek, P.G. (1997) Kinetics of carbon tetrachloride reduction at an oxide- free iron electrode. *Environ. Sci. Technol.*, *31*: 2385.
36. Betterton, E.A.; Arnold, R.G.; Liu, Z.; Diaz, I.; Chen, G. (1997) Electrolytic Systems for the Treatment of Solvent-Contaminated Water. http://www.engg.ksu.edu/hsrc/97Proceed/Remediation2/electrolytic.html.
37. McNab, W.W. Jr.; Ruiz, R. (1998) Palladium-catalyzed reductive dehalogenation of dissolved chlorinated aliphatics using electrolytically-generated hydrogen. *Chemosphere*, *37*: 925–636.
38. Chen, G.; Betterton, E. A.; Arnold, R. G.; Ela, W. P. (2003) Electrolytic reduction of trichloroethylene and chloroform at a Pt- or Pd-coated ceramic cathode. *Journal of Applied Electrochemistry*, *33* (2): 161–169.
39. Wang, J.; Farrell, J. (2005) Feasibility study for reductive destruction of carbon tetrachloride using bare and polymer coated nickel electrodes. *Journal of Applied Electrochemistry*, *35* (3): 243–248.
40. Petersen, M. A.; Sale, T. C.; Reardon, K. F. (2007) Electrolytic trichloroethene degradation using mixed metal oxide coated titanium mesh electrodes. *Chemosphere*, *67* (8): 1573–1581.
41. Li, T.; Farrell, J. (2000) Reductive dechlorination of trichloroethene and carbon tetrachloride using iron and palladized-iron cathodes. *Environ. Sci. Technol.*, *34*: 173.