

PRACTICAL ASPECTS OF IN-SITU ELECTROKINETIC EXTRACTION

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ABSTRACT: Electrokinetic extraction is an emerging technology that can be used to remove contaminants from heterogeneous fine-grained soils in situ. Contaminants in the subsurface are removed by the application of a direct-current electric field across the contaminated soil. The primary contaminant transport and removal mechanisms are electroosmotic advection and ionic migration. However, there are many complex physicochemical reactions occurring simultaneously during the process that may enhance or retard the cleanup process. Nonetheless, the viability of the technology has been established by results obtained from many bench-scale and large-scale laboratory and pilot-scale field experiments performed on various soils. This paper will review the fundamental concepts of the technology and discuss some important practical aspects and design criteria of the technology for field implementations. An example on cost analysis of the technology is also presented to demonstrate the use of various equations presented in the paper and to illustrate the cost-effectiveness of the technology.

INTRODUCTION

Although there are many soil and ground-water remediation technologies available off-the-shelf, not too many are effective and economical for remediation of fine-grained soils. Fine-grained soils pose three significant problems to any potential remediation technology: (1) Low hydraulic conductivity that restricts the fluid volume flow rate; (2) large specific surface area that provides numerous active sites for surface reactions; and (3) many of these reactions are dynamic, pH dependent, reversible, and not very well understood. The low hydraulic conductivity of the fine-grained soil makes it uneconomical and impractical, if not impossible, to apply the pump-and-treat technology. Too low an applied hydraulic gradient will not generate sufficient volume of fluid flow through the soil to clean up the contamination within a reasonable period. Too high an applied hydraulic gradient may fracture the soil hydraulically and create preferential flow paths to accelerate contaminant migration through the subsurface. The large specific surface area of the fine-grained soil provides numerous active sites for surface complexation and sorption of contaminants. These reactions reduce the quantity of contaminant in the dissolved phase and thus reduce the cleanup efficiency of most remediation technologies. If these reactions are permanent and irreversible, they can be used to solidify and stabilize the contaminant in situ. Unfortunately, many of these reactions are dynamic, pH dependent, reversible, and not very well understood. Therefore, the attenuation of contaminant transport provided by these reactions cannot be considered as a permanent contaminant containment mechanism. Thus, there is a growing need for an in-situ technology to remediate contaminated fine-grained soils. The contaminants may be organic or inorganic chemicals. This paper will present an emerging in-situ cleanup technology, i.e., electrokinetic extraction, to meet this end.

FUNDAMENTALS OF ELECTROKINETIC EXTRACTION

Most soil particle surfaces are negatively charged due to isomorphous substitution and presence of broken bonds. Cations (positively charged ions) are adsorbed on the surface to maintain electrical neutrality of the system. When water is added to soil particles, these adsorbed cations dissolve in water and tend to diffuse away from soil particle surfaces to homogenize the cation concentration of the pore fluid. However, the diffusion tendency is counteracted by the electrostatic force exerted on the ions by the negatively charged soil particle surfaces. A diffuse double layer is thus formed. Detailed derivations of the governing equations of the diffuse double layer are presented by Hunter (1981) and Yeung (1992). The diffuse double layer provides a mobile layer of cations in the vicinity of the soil particle surface. When a direct-current (DC) electric field is applied across a wet mass of soil, the mobile cations are moved toward the cathode (negative electrode). If the soil pore is idealized as a capillary, the mobile cations form a concentric shell within the capillary. When this mobile shell is moved toward the cathode, it drags the pore fluid within the shell along resulting in electroosmosis, i.e., a hydraulic flow induced by an electric field. Electroosmosis is a coupled flow as a flow of one type (hydraulic) is generated by a driving force of another type (electrical) (Mitchell 1993).

For practical purposes, electroosmotic fluid volume flow rate is described by an equation analogous to Darcy's law

$$Q = k_e i_e A \quad (1)$$

where Q = fluid volume flow rate (m^3/s); k_e = coefficient of electroosmotic conductivity ($\text{m}^2/\text{V}\cdot\text{s}$); $i_e = \nabla(-\phi)$ = electric field strength (V/m); ϕ = electric potential applied (V); and A = total cross-sectional area perpendicular to the direction of fluid flow (m^2). Values of hydraulic conductivity for different types of soils can vary many orders of magnitude (Freeze and Cherry 1979). However, values of coefficient of electroosmotic conductivity of different soils lie in the narrow range of 1×10^{-9} to 1×10^{-8} $\text{m}^2/\text{V}\cdot\text{s}$ (Acar and Alshawabkeh 1993; Mitchell 1993; Yeung 1994). Therefore, an electric field is a much more effective force in driving fluid through fine-grained soils of low hydraulic conductivity than a hydraulic gradient and vice versa for coarse-grained soils of high hydraulic conductivity. During an electrokinetic extraction process, the applied DC electric field can thus drive an effective electroosmotic advection of contaminant through the soil and/or inject enhancement agents into the contaminated soil.

Moreover, the applied DC electric field can induce ionic migration of contaminants. Ionic migration or electromigration

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is the movement of charged chemical species relative to the movement of pore fluid. Anions (negatively charged ions) are moved toward the anode (positive electrode) and cations (positively charged ions) are moved toward the cathode (negative electrode). The ionic mobilities of ions in free dilute solutions, i.e., the velocities of the ions under the influence of a unit electric field, are in the range of 1×10^{-8} to 1×10^{-7} m²/V-s (Dean 1992). However, the effective ionic mobilities of ions in soils are considerably lower than the lower bound of this range of values as the flow paths in soils are much longer and more tortuous than those in aqueous solutions. Practical ranges of effective ionic mobilities of ions in fine-grained soils are given by Mitchell (1991).

Combining the mechanisms of electroosmotic advection and ionic migration results in the applicability of electrokinetic extraction. The fundamental concept of the technology is depicted in Fig. 1. Cations are moved toward the cathode by the combined actions of electroosmotic advection and ionic migration. The effect of ionic migration of anions may be diminished by that of electroosmotic advection. However, the direction of electroosmotic flow may reverse during a prolonged application of a DC electric field across fine-grained soils (Eykholt and Daniel 1994; West and Stewart 1995; Yeung et al. 1996; Hsu 1997). The phenomenon of reverse electroosmotic flow cannot be described by (1) and is not well understood. Most experimental results obtained to date indicate that ionic migration is the dominant mechanism. However, results obtained by Hsu (1997) on removal of lead and cadmium from Milwhite kaolinite indicate that electroosmotic advection can be a more effective transport mechanism than ionic migration under certain circumstances. (Milwhite kaolinite is a commercially refined product originating from Bryant, Ark.)

The technique has been used successfully to remove >90% of heavy metals (arsenic, cadmium, cobalt, chromium, copper, mercury, nickel, manganese, molybdenum, lead, antimony, and zinc) from clay, peat and argillaceous sand (Lageman et al. 1989); to remove spiked lead from kaolinite (Hamed et al. 1991; Acar and Alshawabkeh 1996); to remove 85–95% of the original concentrations of cadmium, cobalt, nickel, and strontium from laboratory samples prepared from Georgia kaolinite, Na-montmorillonite, and sand-montmorillonite mixture (Pamukcu and Wittle 1992); to remove cadmium from saturated kaolinite (Acar et al. 1994); and to remove sulfate, metals, and other contaminants from high-purity fine quartz sand (Runnells and Wahli 1993). Eykholt (1992) and Eykholt and Daniel (1994) gave similar supporting experimental results on removal of copper from kaolinite and identified other complicating features of the technology. Rødsand et al. (1995) demonstrated the use of acetic acid to depolarize the cathode reaction and an ion-selective membrane to halt the hydroxyl ion migrating from the cathode into the soil. Their experi-

mental results indicate that depolarization of the cathode reaction by acetic acid can enhance electrokinetic extraction of lead while the membrane extraction technique does not enhance the technology as expected. Results on removal of zinc from Georgia kaolinite presented by Hicks and Tondorf (1994) indicate that problems related to isoelectric focusing can be prevented simply by rinsing away the hydroxyl ions generated at the cathode and 95% zinc removal can be achieved. Yeung et al. (1996) demonstrated the possibility of using disodium ethylenediaminetetraacetate (EDTA) solution as an enhancement agent for the removal of lead from a natural kaolinite. Cox et al. (1996) demonstrated the feasibility of using iodine/iodide lixivant to remediate mercury-contaminated soil. The extracted anionic and cationic contaminant ions accumulate at or in the vicinity of the anode and the cathode, respectively. Depending on their concentration and the electrode potential of the metal relative to that of electrolysis of water, metallic ions may deposit on the cathode as metal (Runnells and Wahli 1993). The method is also effective for the removal of organic pollutants such as acetic acid, phenol, gasoline hydrocarbons, and trichloroethylene from contaminated soils (Shapiro et al. 1989a,b; Acar et al. 1992; Bruell et al. 1992; Segall and Bruell 1992; Acar et al. 1993; Shapiro and Probst 1993; Gopinath 1994). The physicochemical soil-contaminant interactions occurring during the process have to be carefully considered as they have a tremendous impact on the cleanup efficiency (Lageman 1993; West and Stewart 1995; Hsu 1997; Yeung et al. 1997). Moreover, enhancing agents injected into the soil to improve the efficiency of the cleanup process will further complicate the soil-contaminant interactions (Eykholt and Daniel 1994; Cox et al. 1996; Yeung et al. 1996; Hsu 1997). Attempts to model the process have been made by Alshawabkeh and Acar (1992, 1996), Eykholt (1992), Datla (1994), Eykholt and Daniel (1994), Datla and Yeung (1994), Hicks and Tondorf (1994), Yeung and Datla (1995), Denisov et al. (1996), Menon (1996), Menon et al. (1996), and others.

PRACTICAL CONSIDERATIONS FOR IN-SITU IMPLEMENTATION

There are many practical aspects of the technology that need to be considered carefully before the technology can be successfully implemented in the field. Some of the prominent practical considerations are discussed in detail as follows.

Soil Type

Results obtained from bench-scale laboratory and pilot-scale field experiments indicate that the technology can be successfully applied to clayey to fine sandy soils. It appears that soil type does not pose any significant limitation on the technology. However, contaminant transport rates and efficiencies depend heavily on soil type and environmental variables. Soils of high water content, high degree of saturation, and low activity provide the most favorable conditions for transport of contaminants by electroosmotic advection and ionic migration. However, soils of high activity, such as illite, montmorillonite, and impure kaolinite, exhibit high acid/base buffer capacity and require excessive acid and/or enhancement agents to desorb and solubilize contaminants sorbed on the soil particle surface before they can be transported through the subsurface and removed (Yeung et al. 1996, 1997; Puppala et al. 1997). The effects of soil mineralogy on removal of chromium from soils by electrokinetics were also investigated by Reddy et al. (1997). Their results indicate that the presence of carbonate and hematite can adversely impact the process.

The technology can be applied to treat heterogeneous soil deposits effectively. Values of hydraulic conductivity in different types of soils within a heterogeneous deposit can vary

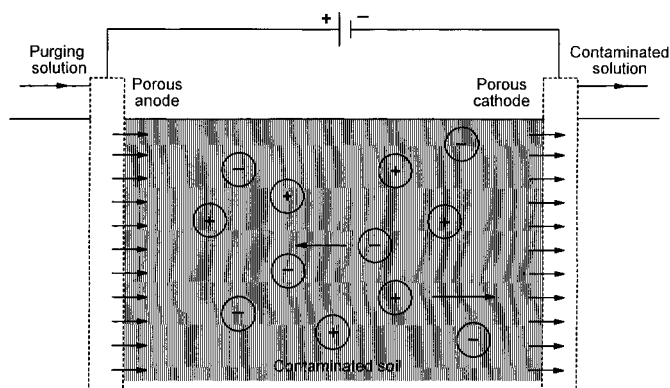


FIG. 1. Concept of In-Situ Electrokinetic Extraction of Contaminants

by many orders of magnitude. For a contaminated soil deposit containing interlayers of sand and clay, typical values of hydraulic conductivity of these strata are 1×10^{-4} and 1×10^{-8} m/s, respectively. If pump-and-treat is used to remediate such a heterogeneous deposit, most of the fluid flow induced will occur in the sandy layer, and the clayey layer will be left untreated practically. Moreover, the high sorptive capacity of the clayey soil for contaminant would further aggravate the problem by retarding contaminant transport. However, values of electrical conductivities of these soils are still within an order of magnitude (Mitchell 1993; Yeung 1994). As a result, the electric field strengths in the different soil layers will be similar when an externally electric potential is applied across the deposit. As the coefficient of electroosmotic conductivity is insensitive to soil type, the electroosmotic fluid volume flow rates in different soil layers will thus be very similar as indicated by (1). As a result, similar electroosmotic advection rates of contaminant transport can be generated in different soil layers within the heterogeneous deposit resulting in a more homogeneous cleanup. The ability to remove contaminants uniformly from a heterogeneous natural deposit is another distinct advantage of the technology.

Contaminants Type and Concentrations

Available experimental data indicate that removal of heavy metals, radionuclides, and selected organics by electrokinetics is feasible. Removal of free-phase nonpolar organics may also be possible if they exist as small bubbles that can be carried by electroosmotic advection (Mitchell 1991). Contaminated colloids may also be removed by the combined effects of electroosmotic advection and electrophoretic migration (Kuo and Papadopoulos 1996). Kuo and Papadopoulos (1996) used latex particles to simulate colloids and observed their movement inside capillaries of different bore diameters under a DC electric field. Their experimental results indicate that both magnitude and direction of the velocity of the particle depend on the magnitude of the electric field applied and bore diameter of the capillary. In fact, the technology can take advantage of colloid-facilitated transport of contaminant. It is anticipated that salts, such as PbO, may dissolve and migrate if an acid front can be generated and advanced through the soil. If an effective acid front cannot be developed by electrokinetic processes probably due to the high acid/base buffer capacity of the soil, enhancement agents can be injected to form soluble complexes (Cox et al. 1996; Yeung et al. 1996; Wong et al. 1997). Solutions of low pH values cannot be injected into the subsurface as they are considered to be hazardous materials by the U.S. EPA regulations. Results obtained by Lageman et al. (1989) indicate that the process can migrate a mixture of different contaminants in soil simultaneously. The technology is thus not selective on the type of contaminant to be removed. Therefore, the type of contaminant does not pose a significant limitation on the technology provided the contaminant does not exist in the sorbed phase on the soil particle surface or as precipitates in the soil pore.

Existing experimental data indicate that removal of Cu(II) with concentrations up to 10,000 $\mu\text{g/g}$ of soil and Pb(II) with concentrations up to 5,000 $\mu\text{g/g}$ is possible. However, a high concentration of ions in the pore fluid will increase the electrical conductivity of the soil and thus reduce the efficiency of electroosmotic fluid flow (Gray and Mitchell 1967; Lockhart 1983a-c). Moreover, the strength of the electric field applied may have to be reduced to prevent excessive power consumption and heat generation during the process. Nonetheless, the concentration of contaminant does not pose any unsurmountable hurdle to the application of the process.

Voltage and Current Levels

The electric current intensities used in most studies are in the order of a few tens of milliamperes per square centimeter. Although a high current intensity can generate more acid and increase the rate of transport to facilitate the contaminant removal process, it increases power consumption tremendously as power consumption is proportional to the square of electric current. An electric current density in the range of 1–10 A/m^2 has been demonstrated to be the most efficient for the process. However, appropriate selection of electric current density and electric field strength depends on the electrochemical properties of the soil to be treated, in particular the electric conductivity. The higher the electric conductivity of the soil is, the higher the required electric current density will need to maintain the electric field strength required. An electric field strength in the order of 50 V/m can be used as an initial estimate for the process. An optimum electric current density or electric field strength should be selected based on soil properties, electrode spacing, and time requirements of the process. Details will be discussed later in this paper.

Effluent Chemistry and Enhancement Scheme

Contaminants can exist in different chemical forms in the subsurface depending on environmental conditions. They can exist as solid precipitates, dissolved solutes in the soil pore fluid and/or ground water, sorbed complexes on the soil particle surface, and/or bonded species on organic matters in the soil. Among these different forms, only dissolved solutes are mobile and removable by electrokinetic extraction and many other remediation technologies. Transformation processes of the contaminant between different chemical forms are contaminant specific, reversible, and dependent on environmental conditions. Nonetheless, most contaminants can be transformed to their dissolved forms. The acidic environment generated at the anode aids in desorption and dissolution of metal contaminants from the soil particle surface. However, the basic environment generated at the cathode can hinder the removal of metal contaminants from the soil. In some cases, it is necessary to inject reagents into the soil to enhance solubilization and transport of metal contaminants. In an acidic environment, some metal ions exist as cations while some exist as anionic complexes. Both chemical forms are soluble and thus can be extracted from the soil by the process.

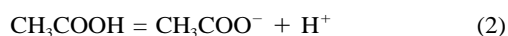
If the metal contaminant exists in a cationic form, such as lead, it will transport toward the cathode under the applied DC electric field. It will eventually accumulate in the catholyte inside the hollow cathode or deposit on the cathode surface and be removed from the subsurface. This mechanism of cationic extraction is very important as the transport of protons across the soil results in a very efficient and cost-effective soil washing. However, some metal contaminants, such as arsenic, will form anionic soluble complexes in an acidic environment. It is thus necessary to accommodate removal of metal as anions at the anode. In both cases, enhancement agents may be needed to control the soil chemistry, and to promote solubilization and transport of the species. Enhancement agents should process these important characteristics as follows: (1) they should not form insoluble salts with the contaminant within the range of pH values expected to develop during the process; (2) they should form soluble complexes with the contaminant that can electromigrate efficiently under a DC electric field; (3) they should be chemically stable over a wide range of pH values; (4) they should have a higher affinity for the contaminant than the soil particle surface; (5) they and the resulting complexes should not have a strong affinity for the soil particle surface; (6) they should not generate toxic residue in the

treated soil; (7) they should not generate an excessive quantity of wastewater or the end products of the treatment process should be amenable to concentration and precipitation after use; (8) they should be cost-effective including reagent cost and treatment costs for the waste collected and/or wastewater generated; (9) they should not induce excessive solubilization of soil minerals or increase the concentrations of any regulated species in the soil pore fluid; and (10) if possible, they should complex with the target species selectively.

Different schemes have been proposed and evaluated to enhance transport and extraction of cationic species under a DC electric field and to prevent the formation of immobile precipitates. A brief discussion of these enhancement alternatives is presented.

Enhancement Agents for Catholyte Neutralization

Weak acids, such as acetic acid, may be introduced at the cathode to neutralize the hydroxyl ions generated by electrolytic reduction of water (Acar and Alshawabkeh 1993; Puppala et al. 1997). However, improper use of some acids in the process may pose a health hazard. For example, the use of hydrochloric acid may pose a health hazard because (1) it may increase the chloride concentration in the groundwater; (2) it may promote the formation of some insoluble chloride salts, e.g., lead chloride, and (3) if it reaches the anode compartment, chlorine gas will be generated by electrolysis. Acetic acid is a weak acid that undergoes partial dissociation in water



There are several advantages in using acetic acid to depolarize the hydroxyl ions generated by the cathodic electrolytic reduction processes: (1) Most metal acetates are highly soluble; (2) the concentration of ions generated by dissociation of the acid is very low due to the high pKa value of acetic acid, and thus the electrical conductivity of the soil will not increase drastically; (3) it is environmentally safe and biodegradable; and (4) acetate ions will prevent the formation of other insoluble salts in the vicinity of the cathode, thus preventing the development of a low electrical conductivity zone and dissipation of excessive energy in the soil near the cathode.

Nafion Membrane Enhancement

Another enhancement for the cationic extraction technique that has been investigated by Electrokinetics (EK) Inc. of Baton Rouge, La., is the use of a Nafion (DuPont, Wilmington, Del.) membrane at the cathode (EK 1995). The purpose of installing a Nafion membrane at the cathode is to prevent or minimize the migration of hydroxyl ions generated at the cathode into the soil. Nafion is a polymer permeable to different cations and polar compounds. The size and properties of these species determine their mobility through the membrane. However, the material is impermeable to anions and nonpolar negatively charged compounds. Moreover, it is insoluble in most solvents and chemically resistant to most strong oxidizing agents and strong bases.

Chelating or Complexing Agents

In some cases, an acid front may not be able to develop by electrokinetic processes because of the high acid/base buffer capacity of the soil and/or reverse electroosmotic flow, i.e., from the cathode toward the anode (Yeung et al. 1996, 1997). Moreover, the advance of an acid front in the subsurface may cause too much dissolution of soil minerals resulting in an excessive release of some of their constituents, such as Al and Si. Under these circumstances, it is necessary to use enhancement agents to solubilize the contaminants.

Chelating or complexing agents, such as citric acid and EDTA, have been demonstrated to be feasible for the extraction of different types of metal contaminants from fine-grained soils (Pamukcu and Wittle 1992; Eykholt and Daniel 1994; Yeung et al. 1996; Wong et al. 1997). In some of these studies, reverse electroosmotic flows were observed in the soils used. The enhancement agents were injected into the soil to compete with the soil particle surface to form negatively charged soluble complexes with the metal contaminants. The dissolved anionic complexes in the pore fluid were transported by electroosmotic advection and electromigration toward the anode where they were collected and extracted. In some other applications, the enhancement agent is simply used to increase the solubility of the contaminant by formation of complexes (Cox et al. 1996). Nonetheless, the choice of enhancement agent is site and contaminant specific. The sorption characteristics of the contaminant on the soil particle surface in the presence of the enhancement agent as a function of the value of pH must be carefully studied if the primary function of the enhancement agent is to desorb the contaminant from the soil particle surface. The presence of the enhancement agent can change the sorption characteristics completely. For example, results obtained by Yeung et al. (1996) indicate that the addition of EDTA promotes sorption of lead and cadmium onto kaolinite particle surfaces in an acidic environment, rendering these metal contaminants immobile. The ability of the enhancement agent to form complexes with the metal contaminant may also be pH dependent. Although the formation of complexes in free solution can be predicted by solving simultaneous chemical equilibrium equations using appropriate equilibrium constants, the behavior of these complexes in the presence of soil at different pH values should also be evaluated experimentally. The concentration of the enhancement agent required must also be carefully determined from these laboratory investigations. An improper choice of enhancement agent may aggravate the contamination problem and make the remediation process much more difficult (Yeung et al. 1996).

Electrode Material, Configuration, and Spacing

Chemically inert and electrically conducting materials such as graphite, coated titanium, or platinum should be used as anode to prevent dissolution of the electrode and generation of undesirable corrosion products in an acidic environment. If necessary, sacrificial electrodes can also be used as anode. Any conductive materials that do not corrode in a basic environment can be used as cathode. Important considerations for the choice of electrode material are (1) electrical conduction properties of the material; (2) availability of the material; (3) ease of fabrication to the form required for the process; (4) ease of installation in the field; and (5) material, fabrication, and installation costs. Regardless of the material selected for the electrode, the electrode has to be installed properly in the field so that it can make good electrical contact with the subsurface. Moreover, the design must make provisions to facilitate exchange of solution with the subsurface through the electrode.

An open electrode configuration allowing exchange of solution between the electrode and the subsurface environment is essential for proper functioning of the process. The electrode should be electrically conductive, chemically inert, porous, and hollow. The hollow in the electrode can facilitate removal of contaminated solution from the subsurface and/or injection of purging solution into the subsurface as shown in Fig. 1. The electrodes can be installed horizontally or vertically. Limited research has been conducted to study the effect of the electrode configuration on the efficiency of electrokinetic extraction. Most bench-scale and large-scale laboratory and pilot-scale field studies on electrokinetic remediation performed to date

have been one-dimensional (1D). Effective and efficient full-scale field application will require an optimal electrode configuration. 1D, two-dimensional (2D), or axisymmetrical electrode configurations may be adopted. Although limited numerical sensitivity analyses have been performed on 2D or axisymmetrical configurations (Renaud and Probst 1987; Jin and Sharma 1991), no experimental data are available to evaluate the field performance of these different electrode configurations.

For a 1D electrode configuration, sheet electrodes can be installed in the field by a procedure similar to the installation of wick drains. The electrodes are practically driven into the soil. Using electrode trenches is another 1D application. However, an approximately 1D electric field can also be obtained by lines of rod electrodes placed in boreholes. It is probably the easiest method to install and the most cost-effective approach for in-situ remediation. However, this configuration may develop spots of inactive (dead) electric field between electrodes of the same polarity. Hexagonal, square, or triangular electrode configurations can be used for 2D field implementation. In a hexagonal electrode configuration, electrodes form honeycomb cells containing a cathode surrounded by six anodes as shown in Fig. 2(a). In a square configuration, electrodes form square cells containing a cathode surrounded by four, eight, or may be even more anodes as shown in Fig. 2(b). Similarly, in a triangular configuration, electrodes form triangles containing one cathode surrounded by three anodes as shown in Fig. 2(c). In these configurations, the cathode is placed at the center and the anodes are placed on the perimeter to maximize the spread of the acidic environment generated by the anodes and to minimize the extent of the basic environment generated by the cathode. These configurations of electrodes generate 2D nonlinear electric fields. Spots of inactive electric field can still develop in these configurations during electrokinetic processing of contaminated soils. However, the areas of these inactive spots are smaller than those developed in the approximate 1D configuration containing parallel lines of anodes and cathodes. In a 1D configuration, the electric current density, i.e., current per unit area, is independent of location. In 2D configurations, however, the electric current density increases linearly with distance toward the cathode. Therefore, the electric field strength also increases linearly with distance toward the cathode.

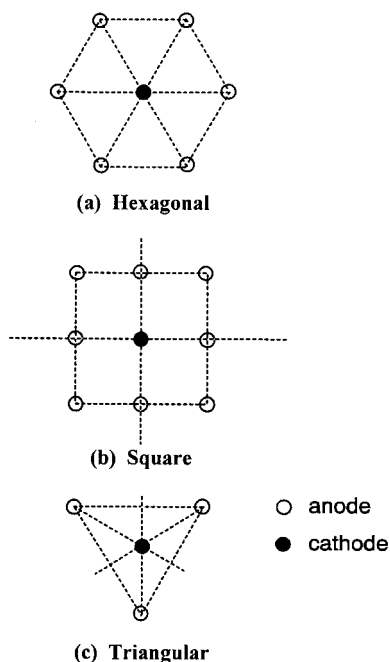


FIG. 2. Examples of 2D Electrode Configurations

Factors affecting the selection of electrode configuration for full-scale field implementation include (1) location and size of any inactive electric field spots that can be developed; (2) number and costs of electrodes per unit area to be treated; and (3) time requirements of the designed remediation process. Factors affecting the selection of electrode spacing include (1) costs; and (2) processing time required.

A larger electrode spacing will reduce the number of boreholes and installation costs, but will increase the processing time required and operation costs. The processing time required is a function of the rate of contaminant transport and electrode spacing. As electroosmotic advection and ionic migration are the prominent transport mechanisms, hydrodynamic dispersion and retardation can be neglected in preliminary analyses. The rate of species transport under an electric field is given by

$$v = (nu\tau + k_e)\nabla(-\phi) \quad (3)$$

where v = rate of species transport assuming the soil is a homogeneous medium (m/s); n = porosity of the soil (dimensionless); u = ionic mobility of the species ($m^2/V\cdot s$); τ = factor that accounts for soil tortuosity and species concentration (dimensionless); and k_e = coefficient of electroosmotic conductivity ($m^2/V\cdot s$). If the spacing between electrodes of opposite polarity is chosen to be L , the time t required for remediation can be estimated by

$$t = \frac{L}{(nu\tau + k_e)\nabla(-\phi)} \quad (4)$$

The typical relationships between the processing time required and electric field strength for different electrode spacings are shown in Fig. 3. These relationships are generated by using the following typical parameters: (1) Ionic mobility of the species u of $5 \times 10^{-8} m^2/V\cdot s$; (2) coefficient of electroosmotic conductivity of the soil k_e of $1 \times 10^{-9} m^2/V\cdot s$; (3) porosity of the soil of 0.4; and (4) tortuosity factor τ of 0.3. Moreover, it is assumed that there is no retardation on the transport of species.

However, it should be noted that these are simplified estimations and the contaminant of interest is assumed to be readily available for transport in the soil pore fluid. This is probably the exception rather than the rule in real-life field implementation of the technology. Heavy metals are usually either sorbed on the soil particle surface or precipitated in the soil pore. Therefore, their transport is retarded by sorption and precipitation. A delaying factor (similar to the retardation factor in advection-dispersion contaminant transport) can be introduced in (4) to account for the extra time required for acid transport, metal desorption and dissolution, etc. If enhance-

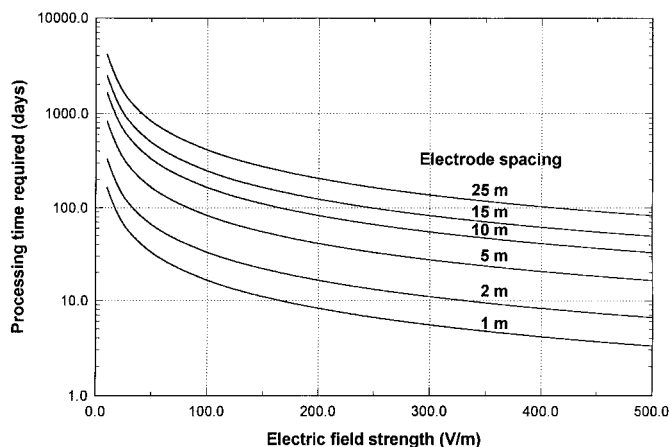


FIG. 3. Processing Time Required versus Electric Field Strength as Function of Electrode Spacing

ment agents are used to solubilize heavy metals, this factor should be modified accordingly. Eq. (4) can then be modified to

$$t = \frac{R_d L}{(nu\tau + k_e)\nabla(-\phi)} \quad (5)$$

where R_d = delaying factor (dimensionless). The value of R_d depends on soil type, pH, and type of contaminant. Sorption retardation factor can be used as an initial estimate of R_d , and it equals unity for nonreactive contaminants.

Energy Expenditure

Total energy expenditure to treat a unit volume of contaminated soil depends on many factors including soil properties, contaminant properties, and electrode configuration and spacing. If the electrical conductivity of the contaminated soil is assumed to be constant throughout the process as a first approximation, the energy expenditure per unit volume of contaminated soil is given by

$$W = \frac{\phi I t}{L} \quad (6)$$

where W = energy expenditure per unit volume of soil (J/m^3); and I = electric current density (A/m^2). Substituting the point form of Ohm's law, i.e., $I = \sigma \nabla(-\phi)$, and (5) into (6)

$$W = \frac{\phi}{\beta} \quad (7)$$

where $\beta = (nu\tau + k_e)/(R_d\sigma)$ [m^3/C or $(m/s)/(A/m^2)$]; and σ = electrical conductivity of the soil (S/m). β is a lumped property of the contaminant and the soil. It represents the rate of transport of a specific species per unit electric current density. It is evident from (7) that the energy expenditure depends on both the soil and contaminant characteristics. A high coefficient of electroosmotic conductivity k_e or a high ionic mobility of the contaminant u will increase the value of β and reduce the energy expenditure W . A high concentration of contaminant or a high ionic strength of the pore fluid will increase the electrical conductivity of the soil σ , reduce the value of β , and thus increase the energy expenditure W . An increase in the delaying factor R_d will also increase energy expenditure. Typical values of β for contaminated fine-grained soils are estimated to be in the range of 1×10^{-8} to $1 \times 10^{-6} m^3/C$. Relationships between energy expenditure and the value of β for different electrode spacings are depicted in Fig. 4.

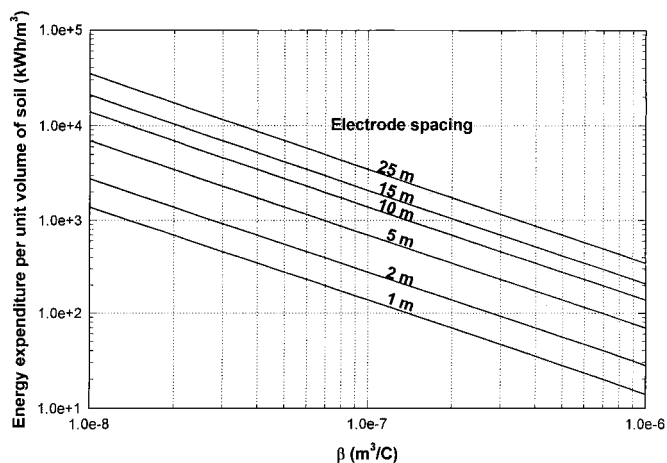


FIG. 4. Relationships between Energy Expenditure and β as Function of Electrode Spacing (Electric Field Strength = 50 V/m)

Costs

Schultz (1997) presented a methodology for estimating costs of electrokinetic remediation. A similar procedure is used in this paper. In general, the total costs for full-scale in-situ implementation of the technology may have five major components: (1) Costs for fabrication and installation of electrodes; (2) cost of electric energy; (3) cost of enhancement agent if necessary; (4) costs of any posttreatment if necessary; and (5) fixed costs. These costs are discussed in detail as follows.

Costs for Fabrication and Installation of Electrodes

The costs of each electrode depend on the material used, complexity of installation, and dimensions. The number of electrodes per unit volume of soil to be treated depends on electrode configuration and spacing. The installation costs depend on the method of installation, depth of the electrodes to be installed, and number of electrodes to be installed. The total costs of electrodes per unit volume of soil to be treated include the material and fabrication costs of the electrodes and their installation costs in the field.

Cost estimates for a 1D electrode configuration are used as an illustration. Lines of electrodes will be installed by drilling boreholes and placing rod electrodes in the boreholes. The spacing between electrodes of the same polarity is taken to be one-third of the spacing between a line of anodes and a line of cathodes, i.e., L , to minimize the number inactive spots developed between electrodes of the same polarity. An area L^2 is considered. Each of the four electrodes at the corners serves four adjacent areas, therefore their total contribution to the area is one electrode. Each of the four electrodes on the perimeter serves two adjacent areas, and their total contribution per area is two electrodes. Therefore, the total number of electrodes for the area of L^2 is three. Consequently, the number of electrodes required per unit plan area of soil is given by

$$N = \frac{F}{L^2} \quad (8)$$

where N = number of electrodes per unit area of soil to be treated; and F = factor depending on electrode configuration. $F = 3$ for the given 1D electrode configuration. The total electrode costs per unit volume of soil to be treated are given by

$$C_{\text{electrode}} = C_1 \frac{F}{L^2} \quad (9)$$

where $C_{\text{electrode}}$ = electrode costs per unit volume of soil to be treated; and C_1 = cost of an electrode to be installed per unit length. C_1 includes the unit costs for material and fabrication of the electrode, drilling and preparation of the borehole, and placement of the electrode. It is evident from (9) that the electrodes' costs decrease with increase in electrode spacing.

Electric Energy Cost

Eq. (7) provides an estimate for energy expenditure per unit volume of the soil treated. Thus, electric energy cost of the treatment process can be estimated by

$$C_{\text{energy}} = \frac{C_2 \phi}{3,600,000 \times \beta} \quad (10)$$

where C_{energy} = electric energy cost per unit volume of soil treated ($\$/m^3$); C_2 = electric energy cost ($\$/kW \cdot h$). Although electrode spacing is not explicitly expressed in (10), its impact on energy expenditure is illustrated in Fig. 4. The larger the electrode spacing is, the higher the voltage and energy expenditure will need to be if a given strength of electric field has to be maintained.

Cost for Enhancement Agent

If the use of enhancement agent becomes necessary, the cost of the chemical should be included. A laboratory investigation should be conducted to evaluate the efficiency of the enhancement agent to remove contaminants from the soil, i.e., the volume of soil that can be remediated by a unit volume of enhancement agent. The cost for enhancement agent can be estimated by

$$C_{\text{chemical}} = \frac{C_3}{\zeta} \quad (11)$$

where C_{chemical} = cost of chemical per unit volume of soil to be treated; C_3 = cost of the chemical per unit volume ($\$/\text{m}^3$); and ζ = remediation efficiency of the chemical. Costs for different chemicals are available commercially.

Costs of Posttreatment

If the effluent from the process required posttreatment or a small portion of the treated soil needs to be removed due to accumulation of a high concentration of contaminant that cannot be extracted, there will be posttreatment costs. These costs are highly site and contaminant specific. They are also dependent on the enhancement agent used in the process. Therefore, they have to be quantified on a case-by-case basis.

Fixed Costs

Fixed costs include mobilization and demobilization costs of various equipment, site preparation, security, progress monitoring, insurance, labor, contingency, and miscellaneous expenses. The equipment will not be consumed in a particular project. However, there are capital, depreciation, or rental costs involved.

Total Costs

The total costs per unit volume of soil to be treated are thus given by

$$C_{\text{total}} = C_{\text{electrode}} + C_{\text{energy}} + C_{\text{chemical}} + C_{\text{posttreatment}} + C_{\text{fixed}} \quad (12)$$

where C_{total} = total costs per unit volume of soil to be treated ($\$/\text{m}^3$); $C_{\text{posttreatment}}$ = posttreatment costs per unit volume of soil to be treated ($\$/\text{m}^3$); and C_{fixed} = fixed costs per unit volume of soil to be treated ($\$/\text{m}^3$).

OPTIMUM ELECTRODE SPACING

Assuming that a 1D electrode configuration is used and posttreatment, chemical, and fixed costs are independent of electrode spacing, the optimum electrode spacing can be obtained by equating the partial derivative of C_{total} with respect to L to zero, i.e.

$$L_{\text{optimum}} = \sqrt{\frac{7,200,000 \times \beta C_1 F}{C_2 \phi}} \quad (13)$$

where L_{optimum} = optimum electrode spacing (m). An estimate of the optimum electrode spacing that minimizes the total costs of the remediation process as a function of the properties of the contaminated soil and electric field strength required is given in (13). If it is necessary to include the processing time required explicitly, (5) can be included in (13) to evaluate the optimum electrode spacing, i.e.

$$L_{\text{optimum}} = \sqrt[4]{\frac{7,200,000 \times \beta^2 \sigma C_1 F t}{C_2}} \quad (14)$$

DESIGN EXAMPLE

An example is given on the estimation of total costs per unit volume of contaminated soil to be treated by electrokinetics.

In this example, a contaminated area of $50 \text{ m} \times 100 \text{ m}$ in plan is assumed. The depth of contamination is assumed to be 5 m. The soil is a saturated silty clay. The porosity, tortuosity, electrical conductivity, and coefficient of electroosmotic conductivity of the soil are determined from preliminary laboratory analyses to be 0.4, 0.3, 0.02 S/m, and $1 \times 10^{-9} \text{ m}^2/\text{V}\cdot\text{s}$, respectively. The ionic mobilities of target contaminants are taken to be $5 \times 10^{-8} \text{ m}^2/\text{V}\cdot\text{s}$. The value of the delaying factor R_d is dependent on soil sorption capacity, soil acid/base buffer capacity, and pore fluid chemistry. It is taken to be 2 in this example. The value of β is thus calculated to be $1.75 \times 10^{-7} \text{ m}^3/\text{C}$.

The mobilization cost of a drilling rig and the labor cost of a two-man operating crew are taken to be \$1,000/day to evaluate C_1 in (9). For boreholes to be drilled to a depth of 5 m without installation of casing and sampling, a continuous flight auger can achieve $\sim 65 \text{ m/day}$. Therefore, the drilling cost is estimated to be \$15/linear m. Costs for fabrication and installation of graphite electrodes are approximately \$5/linear m as the electrodes are reusable. Therefore, C_1 is taken to be \$20/linear m. The electricity cost C_2 is approximately \$0.04/kW·h. If a 1D electrode configuration is used with spacing between the electrodes of the same polarity equal to one-third of anode-cathode spacing, the factor F in (9) equals 3.

There are two limiting factors that need to be considered, i.e., processing time or electric voltage available. If time available is taken to be the limiting factor and the remediation has to be finished in six months, (14) yields an optimum electrode spacing of 3.2 m. The electric field strength required is determined by (5) to be 58.5 V/m. The electric voltage across electrodes of opposite electrodes is thus 187.2 V. The electrode costs per unit volume of soil are given by (9) to be \$5.9. The energy cost per unit volume of soil is given by (10) to be \$11.9. If chemical, posttreatment, and fixed costs are estimated to be \$30 per unit volume of soil, the total costs are \$48 per unit volume of soil. Total costs of the project will thus be \$1,200,000.

If the electric voltage available is the limiting factor, the limiting voltage should be used in (13) to obtain the optimum electrode spacing. The processing time required, energy expenditure, and total costs per unit volume of contaminated soil of the remediation process can then be estimated similarly. If the limiting voltage is taken to be 100 V in this example, the optimum electrode spacing given by (13) is 4.35 m. The electrode costs are \$3.17 per unit volume of soil as given by (9). The energy costs per unit volume of soil is \$6.35 as given by (10). The total costs per unit volume of soil are \$39.52. Total costs of the project are thus reduced to \$988,000. However, it will take more than 20 months to remediate the site as estimated by (5).

SUMMARY

Problems associated with remediation of contaminated heterogeneous fine-grained soils are presented. The fundamentals of an emerging in-situ technology, i.e., electrokinetic extraction, are reviewed. The viability of the technology has been established by many bench-scale and large-scale laboratory experiments and pilot-scale field investigations. Some important practical aspects and design criteria of the process are presented. Different components of costs associated with the process are discussed in detail. These factors must be carefully considered before the technology can be successfully implemented in the field. An example on cost analysis of the technology is presented to illustrate the use of equations presented and cost-effectiveness of the process.

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