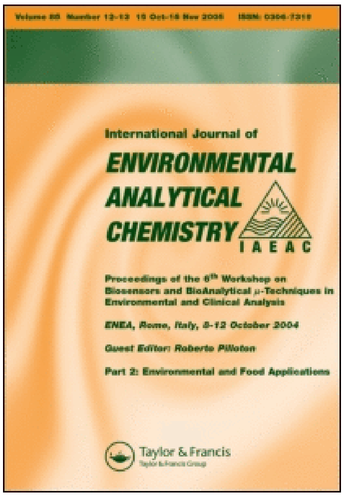

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MODELLING OF DECONTAMINATION RATE IN AN ELECTROKINETIC SOIL PROCESSING

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Modelling of the soil decontamination rate is developed for the case of an electrokinetic remediation run under controlled pH conditions. This model is based on a simple expression of the electrokinetic velocity of ionic species, including some parameters depending on the soil and pollutant species. Laboratory experiments run on kaolinite, using some cations as contaminant models fit well the theoretical calculations.

Keywords: Electroremediation; kaolinite; velocity; decontamination rate; modelling

INTRODUCTION

Most of the existing treatment technologies for the decontamination of contaminated soils are still costly and time consuming, generally not resulting in a complete and real rehabilitation of polluted sites. For example, soil washing, which requires many different treatment stages, is mainly efficient for the sandy fractions of contaminated soils. Methods of confinement (with hydraulic or non-natural solid barriers), solidification, excavation and disposal at a landfill are technologies that only surround and restrain the pollution for as yet an undefined period of time[1].

The present day urgency to get feasible and economical remedial actions for soils have resulted in the development of alternative technologies. The electrokinetic soil processing (also named electroreclamation or electroremediation) is one of the most promising innovative method for the removal of

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pollutants from soils. The significance of this technology appears to be in its projected low operation cost and its potential applicability *in* or *ex situ* to various contamination situations[2]. Several experiments, both at field and laboratory scales, have already been conducted using this method. High degrees of efficiency for a wide range of contaminants have been reported. Ionic and non-ionic species, heavy and light metals[2-6], as well as organic compounds[7-11] can be successfully removed from soil by the electrokinetic soil processing[3, 7-13].

Conceptually the method is quite simple. Between electrodes inserted in the contaminated soil mass, an electric field is applied generating a direct current. Simultaneous flows of fluid, electricity and chemical species then arise. The contaminants mobilized are transported through the porous matrix towards one of the electrodes, to be extracted by adapted collection or separating processes.

The contaminant transport during the electrokinetic soil processing then appears as a multiphenomena process. The pore fluid convection and the transport of the dissolved species within the liquid (due to different transport mechanisms) characterize the contaminant motion. But other phenomena can also arise during the treatment: reactions or interactions can occur at the solid/liquid interface (sorption, dissolution, chemical interactions) at the electrodes and/or in the bulk fluid.

Owing to the simultaneous occurrence of these various phenomena and their complex interactions, the entire process still remains complicated to model. However, models including some combined effects are now proposed[8, 15-17, 20, 21]. But recent studies have emphasized that the theoretical approach and formulation of the complex transport processes and the chemistry involved have yet to be developed[2, 14-19]. Mathematical modelling should facilitate the exploration of a wide range of variations in electroremediation conditions to optimize its performance and develop it into a well-engineered and reasonably predictable process for a variety of field applications.

In the present work, after a rapid description of the different possible electrokinetic phenomena, a theoretical expression for the velocity of charged species in the pore solution during the treatment process is first developed, based on the assumptions that electroosmosis and electromigration are the dominant transport phenomena involved. Additionally, adsorption is integrated to this simple theory through the use of a retardation factor. A one-dimensional model for the percentage of decontamination is then elaborated. With adapted experimental conditions, so as to fit the restricted hypothesis, laboratory experiments are run with kaolinite as soil model, using a pore solution containing one specific ion as contaminant model. Experimental values for the velocities, then for the percentage of decontamination are compared to the values obtained from the theoretical calculations.

THEORETICAL ANALYSIS

The application of an electric field through a porous medium during electroremediation results in three important electrokinetic phenomena.

Electroosmosis is the flow of water induced by an electric field in a porous medium having a charged surface, generally with a net movement of fluid towards the cathode. The soil surface charges (resulting from chemical/physical adsorption, lattice imperfections, isomorphous substitutions...[15, 22]) are usually negative. To balance these charges, mobile cations are in excess near the surface. Therefore, when migrating towards the cathode under the influence of the electric field (see *electromigration*), these cations exert more momentum on the pore fluid than mobile anions do[15]. In practice, electroomosis is described on a macroscopic scale by the empirical relationship established by Casagrande[23]:

$$\mathrm{Qeos} = \mathrm{Ke\ Si_e} \tag{1}$$

where Qeos is the volumic flow rate of water ($\mathrm{cm^3.s^{-1}}$), Ke is the electroosmotic permeability coefficient ($\mathrm{cm^2.s^{-1}.V^{-1}}$), S is the cross sectional area ($\mathrm{cm^2}$, perpendicular to the flow) and i_e is the electrical potential gradient ($\mathrm{V.cm^{-1}}$). The average linear water velocity V_{eos} is then described by

$$V_{eos} = \frac{\mathrm{Qeos}}{\theta S} \text{ or } V_{eos} = \frac{\mathrm{Ke}}{\theta} i_e \tag{2}$$

where θ is the volumetric water content or porosity in saturated conditions.

Electromigration is the movement of charged species under the influence of the electric field $\vec{E}$. This migration, responsible for current conduction in soil water systems[4], depends directly on the value of $\vec{E}$ and the ionic charge number. In aqueous solution, the electromigration velocity, $\vec{V}\mathrm{em}$ is defined by

$$\vec{V}\mathrm{em} = u\,\vec{E}\ (\mathrm{Vem} = u.i_e, \text{ where } i_e \text{ is the electrical gradient}) \tag{3}$$

where u is the electric mobility in free solution defined by

$$u = \frac{zDF}{RT} \tag{4}$$

(z: atomic charge number; D: diffusion coefficient; T: absolute temperature; F: Faraday's constant; R: gas constant).

Cations are drawn towards the negative electrode (cathode) and anions towards the positive one (anode). As the path for electromigration in soils is much longer and more tortuous than in aqueous solutions, the values of electrical mobilities have to be decrease to some effective values, obtained by the empirical expression

$$u^* = \frac{u}{\tau} \quad (5)$$

where τ is an empirical constant called the tortuosity factor ($\tau > 1$)[16, 24].

Electrophoresis is the migration of charged particles within the soil moisture under the influence of an electric field[4, 12]. This phenomenon is often of limited importance in compacted systems[14]. It may play a role when contaminants are adsorbed on them[4], but only becomes significant when surfactants are introduced in the processing fluid or when the technique is used in slurries[25].

Other transport mechanisms can also arise during the electrokinetic treatment.

Diffusion is the movement of chemical species resulting from concentration gradients. The diffusional flux of ions in aqueous solution (expressed by Fick's first law) is directly proportional to the concentration gradient and the diffusion coefficient D in free solution of the specified ion. As for the electromigration the effective diffusion coefficient D* for the diffusional path in soils is obtained by[24].

$$D^* = \frac{D}{\tau} \quad (6)$$

Osmosis is a movement of water resulting from concentration gradients across membranes. Osmosis phenomena in soils are now largely accepted[26, 27], as clays can be considered (generally speaking) as semipermeable membranes[28]. The osmotic flow depends on the osmotic efficiency, which is taken to be zero when the contaminant ions have already migrated into the pore fluid (in this case, the soil cannot be considered as a selective membrane to these ions anymore)[15].

Advection is the movement of fluid resulting from hydraulic potential difference. This can also result in the movement of ions present in the pore fluid (convective transport). It is important to notice that concerning the remediation of fine-grained soils and low permeability clays, the advective flow is small (due to the very low values of their hydraulic conductivity). Under such conditions, an electrical potential gradient will then be much more efficient for moving fluid and contaminants[15, 16, 29].

Other phenomena are inherent to the process. Changes in the soil during electroremediation (pH, various chemical species present in the pore solution, current flow...) can result in different chemical reactions including dissolution, precipitation or complexation reactions. As a consequence, the mobility of a specified pollutant can be modified. Depending on their nature, these reactions can enhance, delay or stop the pollutants removal[25, 30].

The chemistry at the electrodes is governed by electrolysis of water[25]:

At the anode: $2H_2O \rightarrow 4H^+ + 4\bar{e} + O_2(g)$ $E^o = -1{,}23$ V.
At the cathode: $2H_2O + 2\bar{e} \rightarrow H_2(g) + 2\,OH^-$ $E^o = -0{,}83$ V.

The ions produced (H^+ and OH^-) migrate through the soil under the influence of the applied electric field. The pH gradients developed can affect soil properties (i.e., exchange capacity) or chemistry in the bulk fluid. A large number of studies (including some predictive models) have been reported concerning pH gradients and their consequences during the treatment process[4, 9, 13, 35-39].

Finally, interactions between the chemical species and the soil surface are represented by the sorption phenomenon. It includes adsorption, corresponding to the accumulation of contaminant species on the soil solid particles surface and desorption, which is the release of the accumulated pollutants[16]. The mechanisms involved can be of various nature (surface complexation, surface precipitation, ion exchange) depending on the chemical species (size, valence, concentration...), the type of adsorbant (i.e, charge density) as well as pore fluid properties (pH, ionic force...)[33]. Assuming that sorption is fast and reversible, Langmuir or Freundlich isotherms are widely used to describe the phenomenon. The concentration of a chemical species sorbed on the solid phase is supposed to be in equilibrium with that in the adjacent liquid phase. For low concentrations of contaminant the adsorption isotherms are linear. The linear relationship between the solid and the liquid phase concentration is then given by:

$$S_{ad} = K_d.C_s \tag{7}$$

where K_d is a distribution coefficient.

The influence of sorption during contaminants transport is commonly represented by a retardation factor R, defined by

$$R = 1 + \frac{\rho}{\theta} K_d \tag{8}$$

where ρ is the dry density of the soil, K_d the distribution coefficient and θ is the porosity (equal to volumetric water content in saturated soils)[16, 26],40.

SIMPLIFIED THEORY

For simplicity we first assume that:

- the laws governing dilute solutions apply.
- the electrical potential gradient is constant with respect to time and space.
- delaying reactions (precipitation, dissolution, complexation, sorption) in the pore solution or at the solid/liquid interface are neglected.

- hydraulic and concentrations gradients have negligible influence compared with electrical gradients (no diffusion, osmotic, nor advective flow).
- the soil surface is negatively charged.

The transport of charged contaminant species is then only due to electromigration and convection by electroosmosis. The global velocity of an ion i, V^i_{theo}, we name *electrokinetic velocity*, is then the combination of the ionic velocity by electromigration (V^i_{em}) and the average linear fluid velocity by electroosmosis (V_{eos}) [symbols used for velocities represent absolute values]. As both cations and water (by electroosmosis) are drawn towards the cathode, the removal of cationic species is enhanced and their electrokinetic velocity is described by:

$$V^c_{theo} = V^c_{em} + V_{eos} \qquad (9)$$

whereas the anions migration towards the anode is diminished by the electroosmotic flow, which yields to:

$$V^a_{theo} = V^a_{em} - V_{eos} \qquad (10)$$

Finally, using expressions [9] and [10], we get a simple expression (depending on the sign of the ion of interest) for the average linear velocity of a charged species i when electroosmosis and electromigration are the only mechanisms occurring during the transport process:

$$V^i_{theo} = \left(\frac{|u^*|}{\tau} \pm \frac{Ke}{\theta}\right) i_e \qquad (11)$$

This theoretical value V^i_{theo} should help to detect any additional phenomenon, as any difference to this reference value would give evidence that electroosmosis and electromigration are not the two only transport mechanisms.

One of the additional phenomena we choose to study is adsorption, introducing the retardation factor R to represent its influence on the process. Additional hypothesis are:

- S_{ad}, the concentration of adsorbed contaminants (mol/g) and C_s, the concentration of contaminants in the pore fluid (mol.cm^{-3}) are homogeneous all along the soil sample.
- Adsorption is fast and reversible. Contaminant concentrations are low enough to be on the linear part of the adsorption isotherms.

For the conservation of species we write:

$$n_{ad} + n_s = n^o_{ad} - \int_o^t Q(t)dt, \qquad (12)$$

where:

- $\int_0^t Q(t)dt$ is the total quantity of contaminant removed at time t (mole). Q(t) is the outlet molar flow rate of contaminant species ($mol.s^{-1}$) defined by

$$Q(t) = \theta C_s(t)\bar{V}S, \tag{13}$$

where $\bar{V}$ ($cm.s^{-1}$) represent the average linear velocity of the pollutant (other parameters already defined).
- n_{ad} is the quantity of contaminant still adsorbed on the sample at time t (mole).
- n_s is the quantity of contaminant present in the total pore solution at time t (mole). $n_s = C_s\theta\vartheta$, where ϑ is the bulk volume of saturated sample (cm^3), i.e., $\vartheta = SL$, with L the total length of the sample (cm) and $\theta\vartheta$ the total pore fluid volume (cm^3).
- n_{ad}^o is the initial quantity of contaminant adsorbed on the entire soil sample (mole).

From [7], [8], [12] and [13], we get:

$$S\ L\ R\frac{Q(t)}{\bar{V}} = n_{ad}^o - \int_0^t Q(t)dt \tag{14}$$

The decontamination ratio τ_d (ratio between pollutants removed and initially present in the sample) is defined by:

$$\tau_d(t) = \frac{1}{n_{ad}^o}\int_0^t Q(t)dt \tag{15}$$

Combining relations [14] and [15], the expression obtained is derived and gives:

$$\frac{d\tau_d(t)}{1 - \tau_d(t)} = \frac{V\ S\ dt}{S\ L\ R} = \frac{V}{L\ R}dt \tag{16}$$

Supposing that the average linear velocity $\bar{V}$ corresponds to the electrokinetic velocity V_{theo}^i (under the defined restricted hypothesis and conditions), we get a final expression for $\%\tau_d(t)$ [$\%\tau_d(t) = 100\tau_d(t)$], the percentage of decontamination of a charged pollutant i versus time:

$$\%\tau_d = 100\left[1 - \exp\left(-\frac{V_{theo}^i \cdot t}{R\ L}\right)\right]. \tag{17}$$

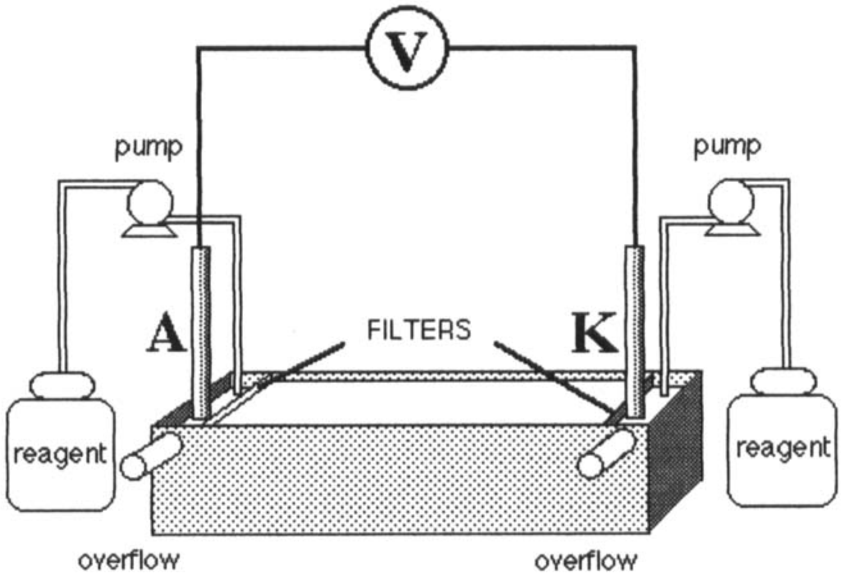

FIGURE 1 Laboratory test cell (A: anode K: cathode V: electric power supply) Dimensions: 50 cm × 4 cm × 4 cm

EXPERIMENTAL

The experimental apparatus is illustrated in Figure 1. The sample is confined in the central part of the PVC cell, separated from the two end chambers by filters of polyamide fibers. A constant voltage is applied between the platinum electrodes located at each end of the cell and immersed in its chamber solution. In order to control the pH throughout the cell, an alkaline solution is continuously added to the anodic chamber, an acidic solution to the cathodic chamber (supplied by a peristaltic pump), to neutralise respectively, the H^+ and OH^- ions produced by the electrolysis of water. Moreover, the continuous supply of fluid at the anode permits a continuous electroosmotic flow and avoids the soil dewatering. Overflow systems allow to collect the effluents at both ends of the cell. During the experiments, measurements are made of the electric field, current density, pH, volumetric flow rate and chemical composition of the effluents.

At the end of each run, the sample is divided into 12 sections and analysed for its pH (by inserting a combination pH electrode directly in the kaolinite sample), water content and average adsorbed contaminant concentration (S_{ad}) after mineralisation of the solid fraction. The pore fluid fraction, separated by centrifugation, is also analysed for its pH, conductivity and contaminant concentration (C_s).

Kaolinite (obtained commercially: extra pure, low bacteria content, MERCK) was chosen as soil model because of its low activity (low adsorption capacity) and relatively high electroosmotic water transport efficiency, as well as a relatively low hydraulic permeability[8, 29].

The buffering solutions used to control the pH consist of ethanoic acid injected in the cathodic compartment to neutralize the produced OH^-, and of ethanoate solution (at the same concentration) injected in the anodic compartment to neutralize the produced H^+. (pKa[CH_3COOH/CH_3COO^-] = 4,8). Ethanoic buffer was chosen with application to actual *in situ* situation in mind, as it is a biodegradable non toxic reagent.

To study the electrokinetic velocity, V_{theo}, different ions were tested as contaminant model: Na^+ (exp. n°1), K^+ (exp. n°2), Ca^{2+} (exp. n°3) and NO_3^- (exp. n°4). Decontamination experiments were conducted with Sr^{2+} (exp. n°5 and n°6) as pollutant model, as this contaminant is of increasing interest in environmental problems, by its implication in nuclear wastes.

The sample is prepared by mixing 500 g of the commercial (for velocities experiments) or artificially contaminated (for decontamination experiments) kaolinite with 400 cm^3 of a buffer solution constituted of ethanoic acid and sodium ethanoate at the concentration of 10^{-2} mol.L^{-1} (Sr^{2+} spike procedure is described elsewhere[36]. The final artificially contaminated kaolinite contains 62 mg of Sr^{2+} per kg of kaolinite).

A constant voltage of 50 V is applied across the cell, corresponding to a theoretical electrical potential gradient of 1 V.cm^{-1}. Continuous flow rates of 30 cm^3.h^{-1} of CH_3COONa (10^{-2} mol.L^{-1}) and CH_3COOH (10^{-2} mol.L^{-1}) are introduced respectively at the anode and at the cathode. This constitute the only step for the decontamination experiments, whereas a second stage is carried out for the study of the electrokinetic velocity. In this last case, when steady state conditions are obtained, i.e., inlet and outlet sodium molar fluxes are equal, sodium is replaced by another cation of interest in the introduced solution (at the anode, as the cations move towards the cathode). The concentration and volumic flow rate remain the same (the procedure used for the nitrate ions is described elsewhere[36]). Continuous flows of ethanoic acid at the cathode and of ethanoate solution (containing now the ion of interest) at the anode are maintained until (after a period of ionic exchange on kaolinite) new steady state conditions again. Such conditions should guarantee that delaying phenomenon are of negligeable influence at the end of the run.

Chemical composition of the various effluents is determined all throughout the run, using a CORNING flame photometer 410 for Na^+ and K^+, atomic absorption spectrometry on a IL 451 flame spectrometer for Ca^{2+} and Sr^{2+} and an Orion Research ionometer for NO_3^-.

RESULTS AND DISCUSSION

Electrokinetic velocity, V_{theo}

Verification of the fundamental assumptions at steady state conditions.

The ions investigated were chosen to avoid any precipitation/dissolution or complexation reactions at the working pH (pH range: 4-6). They are totally dissolved in the pore fluid. The overflow systems, combined to the low hydraulic permeability of kaolinite, assure that hydraulic head difference have negligeable effects compared to electrical potential gradients.

For a given ionic species, the electrical ionic mobility is generally at least one order of magnitude higher than the diffusion coefficient (values of u and D from the literature)[25]. Therefore electromigration is a major contributing component to the total ionic transport. In such conditions, only the development of very important chemical gradients can make the diffusional transport competitive to electromigration[25].

Each run was stopped after steady state conditions for the ionic flow rates were obtained (Figure 2). The ionic concentration in the pore fluid was then quite

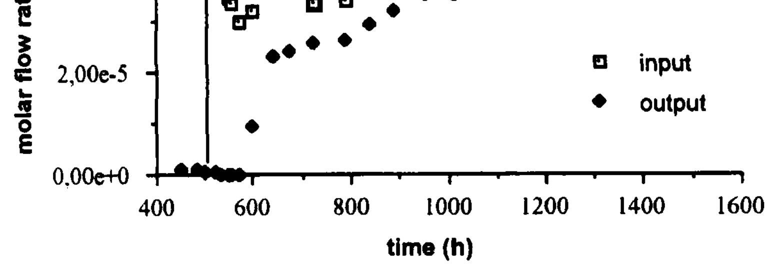

FIGURE 2 Input and output calcium flow rates until steady state conditions (exp. n°3)

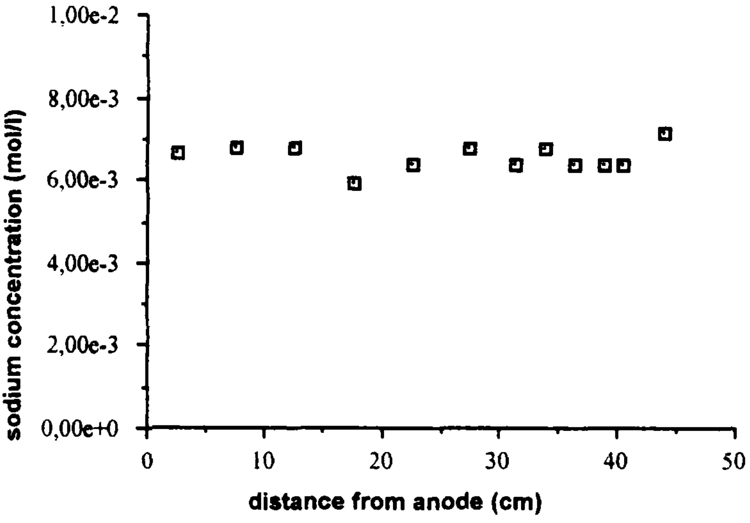

FIGURE 3 Final sodium concentration in the pore solution (exp. n°1)

constant across the sample (Figure 3). The chemical gradients then observed were low enough to neglect diffusion transport. For each run, the possible diffusional flux represented less than 1% of the electrokinetic flux (electro-osmosis + electromigration) under the developed conditions.

A regular electrical potential gradient, around $1 V.cm^{-1}$, was obtained across the sample for each experiment (i.e., Figure 4). This is correlated to the good control of pH[36], which is also regular with respect to time and space in all experiments (Figure 5).

Velocity Results

Velocity values are then calculated using expression [10]:

$$V^{i}_{theo} = (\frac{|u^{*}|}{\tau} \pm \frac{Ke}{\theta})\, i_{e}.$$

Ke is calculated from the experimental evaluation of the electroosmotic flow rate, using relation [1]. u* is obtained, by relation [5], with values from the literature for the electrical mobility u and a value of 1,24 for the tortuosity of kaolinite, as previously reported[10]. Considering that the kaolinite is saturated, θ is defined by,

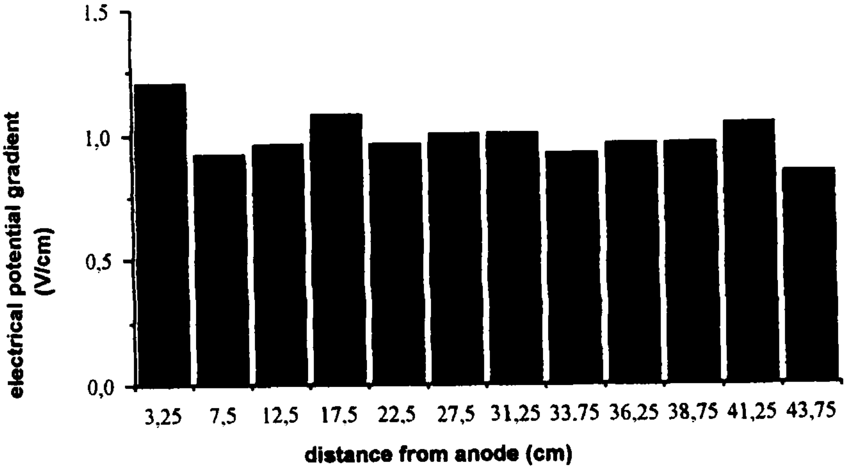

FIGURE 4 Electrical gradients across the sample (exp. n°3)

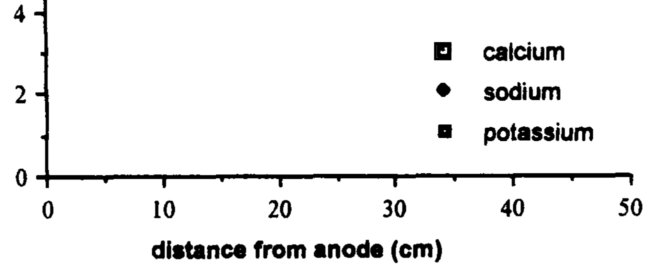

FIGURE 5 pH profile across the kaolinite sample (exp. n°1, 2 and 3)

TABLE I Theoretical and experimental velocities

n° specified ion	[a]u ($cm^2.s^{-1}V^{-1}$)	Ke ($cm^2.s^{-1}V^{-1}$)	[b]i_e ($V.cm^{-1}$)	V_{theo} (cm/day)	V_{exp} (cm/day)
1 (Na^+)	51.10^{-5}	10^{-4}	0,94	35	36
2 (K^+)	67.10^{-5}	$7,3.10^{-5}$	1,04	59	63
3 (Ca^{2+})	57.10^{-5}	$6,1.10^{-5}$	0,99	30	31
4 (NO_3^-)	46.10^{-5}	$5,95.10^{-5}$	1,24	34	38

[a]: the ionic mobility u is given at run temperature [b]: average value of i_e (on the whole sample) are given

$$\theta = \frac{\text{volume of pore fluid}}{\text{total volume}} \tag{18}$$

which yields to $\theta = 0{,}672$ in our conditions.

i_e is the measured electrical potential gradient. As we consider a monodimensional system, i_e is obtained from electrical potential (EP) measurements by

$$i_e = \frac{\Delta EP}{\Delta x} = \frac{EP_{x2} - EP_{x1}}{x2 - x1} \tag{19}$$

where xi (cm) is the distance from the anode and EP_{xi} (V) is the measured EP at xi.

Then the values of V_{theo} are compared to experimental values of velocity, V_{exp}, calculated by

$$V_{exp} = \frac{Q}{\theta\, S\, C_s}, \tag{20}$$

with Q, θ, C_s, S defined above.

A good agreement is obtained between theoretical and experimental values for each experiments (Table I). The predictive expression for the ionic velocity under the specified conditions is therefore validated. It is then introduced in the expression of decontamination ratio as the average linear velocity value $\bar{V} = V^i_{theo}$ to get the final expression [17] of the percentage of decontamination.

Percentage of Decontamination

Batch experiments for the preliminary studies of adsorption of Sr^{2+} on kaolinite are presented elsewhere[36]. Linear isotherms were obtained for the concentration range studied. The influence of pH was studied and a simple relationship was established for K_d versus pH [$\log K_d = f(pH) = apH + b$, with a, b constants[36]]. It allowed to easily predict, with only a simple determination of the pH of the medium, the theoretical retardation factor R, using[8]:

$$R = 1 + \frac{\rho}{\theta} f(pH).$$

Chemical composition of the pore fluid and of the mineralised kaolinite, determined at the end of run n°5, showed that S_{ad}, then C_s which is in equilibrium with S_{ad}, were actually homogeneous across the cell (Figure 6) [The higher values observed in the last part of the sample were correlated to some accidental pH variations in this region[36]]. Short experiments with identical conditions were run to obtain an average value of Ke and i_e in order to determine the electrokinetic velocity of Sr^{2+} (by using expression [11]).

The values of calculated retardation factor R and velocity $V_{theo}^{Sr^{2+}}$ were then introduced in expression [17] to get theoretical evaluation of $\%\tau_d$ versus time. It can be seen on Figure 7 and Table II that they compare very well with the experimental values $\%\tau_d^{exp}$, defined by

$$\%\tau_d^{exp} = 100 \frac{n_r(t)}{n_{ad}^{o}} \qquad (21)$$

where $n_r(t)$ is the quantity of Sr^{2+} (mole) actually removed (and collected) at time t, calculated by $n_r(t) = \Sigma\, Q(t).t$.

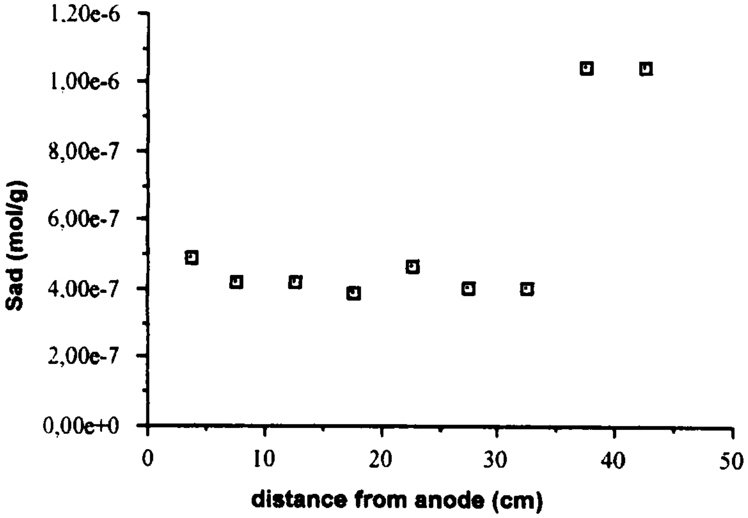

FIGURE 6 Final strontium adsorbed on the kaolinite (exp. n°5)

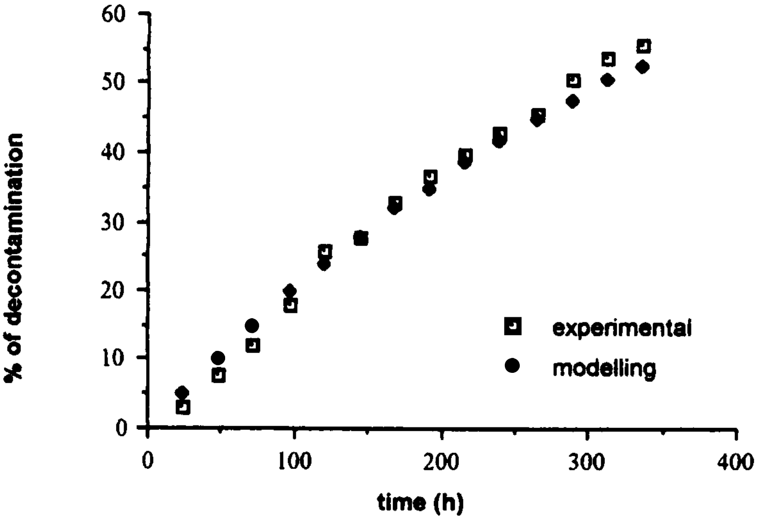

FIGURE 7 Percentage of decontamination of strontium versus time (exp. n°5)

TABLE II Experimental and theoretical percentages of decontamination (exp. n°5)

time(h)	Experimental values of $\%\tau_d$	Theoretical values of $\%\tau_d$
24	3	5
48	8	10
72	12	15
76	18	20
120	26	24
144	28	28
168	33	32
192	37	35
216	40	39
240	43	42
264	46	45
288	51	48
312	54	51

Another decontamination experiment (exp. n°6) run till quite complete decontamination (95% of strontium removed) was carried out during 45 days. It gave similar results with a total agreement between experimental and theoretical values[36]. The simple expression [17] is therefore a good predictive tool for the percentage of decontamination versus time.

CONCLUSION

In this paper, we developed two one-dimensional mathematical models for the electroremediation of low hydraulic permeability soils contaminated with charged species.

With the assumption that electromigration and electroosmosis were the dominant mechanisms, we first developed a simple predictive expression for the average linear velocity of ions present in the pore solution. Due to adapted experimental methodology (i.e., successful control of pH and electrical potential gradients) a good control of the phenomena was achieved during the tests run on kaolinite as soil model, using one ion as pollutant model. The pH control achieved at both ends (anode and cathode) created a suitable environment for the specified ion to remain in the solution as single species so that realistic predictions of velocities were made.

To complete this first model, adsorption was integrated in the theoretical development as a delaying phenomenon. An expression for the percentage of decontamination was then elaborated, based on the ionic velocity and the retardation factor. Predictions of decontamination modelling compare excellently with the results of the one-dimensional tests realised on Sr^{2+} spiked kaolinite under controlled pH conditions. According to the model, strontium ions are ultimately removed from the kaolinite.

This good agreement between calculated and experimental results shows that this approach is a valid step towards a better understanding of the physics and chemistry involved during the treatment process. However further developments are still required to include other phenomena such as precipitation, complexation, diffusion...

Different uses of this model are possible, i.e., information can be obtained on the approximative treatment time necessary to achieve an acceptable percentage of decontamination. On this basis the power consumption and final cost required can also be evaluated.

This model could already constitute a predictive tool for different contamination situations. With only few preliminary studies, in order to determine average values of pH, Ke, R and other specific soil characteristics as porosity, tortuosity, the presented expression can easily be applied.

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