Removal of Cesium and Cobalt within Soil around TRIGA Reactor
by Electrokinetic method
동전기적방법을 이용한 TRIGA 연구로 주변 토양내의 세슘과 코발트 제거

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Abstract

The characteristics of cesium and cobalt removal from soil around the TRIGA reactor using the electrokinetic method were analyzed and a device to restrain the pH increase in the soil column was suggested. When a NaCl solution was used as the electrolyte to raise the electric field strength, a precipitate was formed in the cathode in the soil column, resulting in a low removal efficiency. Thus, an acetate buffer solution (compound solution of CH_3COONa and CH_3COOH) was injected into the soil column and acetic acid was periodically injected into the cathode reservoir to restrain any pH increase. Many Cs^{2+} and Co^{2+} ions were transferred by electromigration rather than electroosmosis during the initial remediation period, and no precipitate was formed in the soil column. 96% of the total amount of cesium in the soil column was removed after 5.9 days, while 94% of the total amount of cobalt was removed. Furthermore, the residual concentrations predicted by the developed model were similar to those obtained by experiment.

Key Words : Electrokinetic method, Acetate buffer, Soil around TRIGA, Cesium, Cobalt

요 약

동전기적방법을 사용하여 TRIGA 연구용 원자로 주변 세슘 및 코발트제거 특성을 분석하고 토양컬럼 내의 pH 상승을 제거할 수 있는 방안을 제시했다. 전기장을 높이기 위해 NaCl 용액이 전해질로 사용되었을 때, 토양컬럼 음극주변에 침전물이 형성되어 낮은 제거효율을 나타냈다. 그래서, pH 의 상승을 억제하기 위해 초산완충액을 토양컬럼에 주입하고, 초산을 주기적으로 음극저수조에 주입했다. 초기제염기간 동안 전기삼투보다는 전기이동에 의해 많은 세슘과 코발트가 제거되었다. 토양컬럼 내의 총 세슘 중 96%가 5.9 일 동안 제거되었고, 총 코발트 중 94%가 제거되었

다. 또한, 개발된 모델에 의한 시뮬레이션 결과는 실험결과와 거의 일치했다.

중심단어 : 동전기적방법, 초산완충액, TRIGA 주변 토양, 세슘, 코발트

I. Introduction

South Korea has many nuclear power plants and two research reactors. As such, the soil around the nuclear facilities can be contaminated with radionuclides over the long-term operation of these facilities. the Korea Atomic Energy Research Institute (KAERI) had removed the soil contaminated with radionuclides from around the TRIGA research reactor in 1987, had placed it in 4000 sets of 200L-wastedrums, and has been storing the drums in a radioactive waste storage in KAERI since 1987. The contamination characteristics of TRIGA soil are shown in Table 1. The smaller the size of the soil particles is, the higher the radioactive concentration in the soil particles is. Plus the contamination concentrations of ^{137}Cs are higher than those of ^{60}Co. However, the radioactivities of ^{137}Cs and ^{60}Co in parts of the contaminated soil have been found to be higher than the restoration target (400 Bq/kg). Therefore, it is necessary to develop a technology for decontaminating the TRIGA soil contaminated with radionuclides.

During the past decade, the necessity of developing economical techniques for decontaminating large areas of contaminated soil has grown. Existing technology, such as soil washing and soil flushing, is only effective in decontaminating the soils with a high hydraulic conductivity. Yet, electrokinetic remediation offers the possibility of in-situ and ex-situ remediation of contaminants in cases where conventional techniques are unfeasible. In electric field restoration, pairs of electrodes are placed in the contaminated soil and a direct current potential is applied across them. The contaminants are then transported under the action of the electric field to an electrode reservoir from which they can be brought to the surface. Unlike soil flushing, the electrokinetic method is effective in soils with a low or variable permeability and does not disperse the contaminants outside the treatment zone.

The ^{137}Cs and ^{60}Co in the large particles (>0.063mm) of TRIGA soil can be removed by the soil washing method, yet, the ^{137}Cs and ^{60}Co in the minute particles (<0.063mm) are hardly removed by this method. In particular, the removal efficiency of ^{137}Cs from minute particles (<0.063mm) by soil washing is extremely low. Accordingly, the current study applied the electrokinetic method to the removal of ^{137}Cs and ^{60}Co from minute particles of soil and investigated the optimum experimental conditions for maximizing the removal efficiency of ^{137}Cs and ^{60}Co.

Heavy metals incorporated into soils take different chemical forms, including a dissolved ionic form, electrostatically absorbed form, and surface complex form. In electrokinetic remediation, the pH of the cathode side in the soil column increases due to the production of OH^- in the cathode electrode. As such, a hydroxide precipitate forms in the cathode side and the removal efficiency of the contaminants is reduced. Li. et al. reported that a decrease in both conductivity and metal precipitation can be avoided by injecting a 0.01 M KNO$_3$ solution into the soil or by inserting a cation selective membrane between the cathode and the soil to stop

Table 1. Contamination characteristics of TRIGA soil

Radiation dose rate of waste drum surface	Soil Particle Size	Volume(%)	Co-60(Bq/kg)	Cs-137(Bq/kg)
above 0.05mR/h (7%)	>1.0mm	28.3	38.9-233.7	1.5-886.0
	0.063-1.0mm	61.2	155.2-1079.3	16.3-6700.2
	<0.063mm	10.5	480.5-6645.0	47.1-19547.0
0.02-0.05mR/h (60%)	>1.0mm	48.5	2.6-24.0	0.3-35.1
	0.063-1.0mm	46.5	6.3-217.8	13.9-436.7
	<0.063mm	5.0	31.1-316.4	287.9-1663.7
below 0.02mR/h (33%)	>1.0mm	52.4	5.3-15.2	2.5-7.0
	0.063-1.0mm	43.1	14.1-69.1	25.4-53.0
	<0.063mm	4.5	56.3-204.7	85.6-377.2

the advancement of the hydroxyl front toward the anode. The electrolyte pH is then controlled by the addition of 0.05 M KOH at the anode and 0.05 M HCl at cathode. Hyunho Lee and Jiwon Yang also developed a new method of controlling the electrolyte pH using a circulation system in electrokinetic soil remediation. Plus, to avoid the precipitation of copper hydroxide in a sand column, Jiwei Yu and Ivara Neretnieks inserted a conductive solution between the sand and the cathode. The removal rates of heavy metals from soil have also been increased with the addition of salts of competing noncontaminant cations, soil acidification, and catholyte drainage. According to the results of previous experiments, after the discharge of about 3 pore volumes of effluent from a soil column by electrokinetic remediation, more than 90% of the cobalt in the soil column is removed. Thus, in the case of soil mixed with a buffer solution in a soil column, it is predicted that the pH in the column will remain constant and any pH increase will be restrained.

In the current study, TRIGA soil (<0.063mm) was artificially contaminated by a 0.01M Cs^{2+} and Co^{2+} solution, and the experiment for removing Cs^{2+} and Co^{2+} from the soil by the electrokinetic method was executed. An acetate buffer solution was mixed with the soil in the column to saturate the soil and restrain any pH increase in the column. Acetic acid was also periodically injected into the cathode reservoir to prevent a pH increase. The effect of the injection of an acetate buffer and acetic acid for removing the Cs^{2+} and Co^{2+} from the TRIGA soil was then analyzed. Finally, a numerical model was developed to simulate the residual concentrations after soil remediation, which was verified by comparing the model values with the experimental ones.

II. Theorem and Modeling

1. Transport mechanisms

The electric field drives the contaminants toward the electrodes by two mechanisms. The first is electroosmosis, an electrokinetic phenomenon in which the saturating liquid and dissolved substances flow toward an electrode. The electoosmotic flow rate is proportional to the product of the applied electric field

strength and the ζ potential at the soil-liquid interface. The value of the ζ potential depends on the soil properties as well as the ionic strength and pH of the saturating liquid. A typical ζ potential is about −10 to − 100 V, and in a 100Vcm^{-1} electric field, water will flow toward the cathode at a velocity of around 1-10 cm min^{-1}. The second mechanism is electromigration in which charged ions move in the electric field strength, the charge on the ion, and the mobility of ion. The transport of charged ions by electromigration is generally more rapid than by electroosmosis. In cases where the ionic strength is high due to the presence of inorganic contaminants, the ζ potential is small and transport occurs mainly by electromigration.

The contribution of each transport to the species flux is derived from the corresponding phenomenological relationship and extended to an isotropic porous medium using a vadose model. The bulk liquid velocity, u_c, in cm min^{-1} is described as the sum of the contribution due to electroosmosis, u_{eo}, in cm min^{-1} and due to any pressure gradient, u_h, in cm min^{-1}. The electroosmotic velocity is calculated from the Helmholtz-Smoluchowski relation

$$u_{eo} = \frac{\varepsilon \zeta}{\tau^2 \mu} \nabla \phi \equiv k_e \nabla \phi \qquad (1)$$

where u_{eo} is the pore velocity due to electroosmosis in cm min^{-1}, ε the permittivity of the pore liquid in F cm^{-1}, ζ the zeta potential of the soil in V, τ a nondimensional tortuosity factor that accounts for the capillary path and is greater than 1, μ the viscosity of the pore liquid in Pa·min, ϕ the electrostatic potential in V, and k_e in cm^2 (V.min)$^{-1}$ an effective electroosmotic permeabililty coefficient used for convenience in the numerical formulation of the model.

The contribution of the pressure gradient is calculated from Darcy's Law

$$u_h = -\frac{k_h}{n\mu} \nabla p \equiv k'_h \nabla p \qquad (2)$$

where u_h is the pore velocity in cm min^{-1} resulting from the pressure gradient, k_h the hydraulic permeability of the medium in cm^2, n the porosity, and p the pressure in Pa. Again, an effective permeability coefficient k'_h in cm^2 (Pa·min)$^{-1}$ is defined for convenience. For an individual species, the electromigration velocity, u_{em}, in cm min^{-1} is given by

$$u_{em} = -\nu zF \frac{\nabla \phi}{\tau^2} \equiv k_m \nabla \phi \qquad (3)$$

where the migration velocity is relative to the pore liquid, v is the mobility defined as the velocity resulting from a unit force per mole, z the charge number, F Faraday's constant, and k_e an effective mobility coefficient in cm^2 (V·min)$^{-1}$.

Finally, the contribution of diffusion to the species flux for dilute solutions is given by Fick's law

$$j^d = -\frac{D}{\tau^2} \nabla C \qquad (4)$$

where j^d is the molar flux due to diffusion in mol(cm^2·min)$^{-1}$, C the molar concentration in the pore liquid in mol cm^{-3}, and D the diffusion coefficient in cm^{2}min^{-1}. Superposing the different mass-transfer mechanisms, the total flux of an individual species is given by

$$j = (u_{eo} + u_h + u_{em})C - \frac{D}{\tau^2}\nabla C \qquad (5)$$

or in terms of the gradient of pressure, electrostatic potential, and concentration by

$$j = [(k_e + k_m)\nabla\phi + k_h\nabla p]C - \frac{D}{\tau^2}\nabla C \qquad (6)$$

where j is the molar flux of species per unit pore area in $mol(cm^2 \cdot min)^{-1}$. The coefficients k_e, k_m, k_h, and $D_i\tau^{-2}$ may vary in space and time.

2. Change equation and charge balance

The equation for concentration change is derived from the application of the conservation of mass in a control volume of porous medium, which contains both the liquid and solid phases. Neglecting transport of adsorbed species, applying Gauss's divergence theorem, and taking the limit of infinitesimal control volume leads to

$$\frac{\partial(C + C^a)}{\partial t} + \nabla \cdot j = R \qquad (7)$$

C^a in mol cm^{-3} is the molar concentration in the adsorbed phase, and R is the net volumetric rates of production of chemical species due to chemical reactions.

But since the total mass of each element is conserved

$$R = 0 \qquad (8)$$

Then, Eq.7 can be rewritten as follows.

$$\frac{\partial(C + C^a)}{\partial t} + \nabla \cdot ((u_{eo} + u_h + u_{em})C - \frac{D}{\tau^2}\nabla C) = 0 \qquad (9)$$

The molar concentration of an individual species in the adsorbed phase is demonstrated by

$$C^a = \frac{\rho}{n}K_d C \qquad (10)$$

where, K_d is the equilibrium sorption coefficient.

Substituting the pore velocity due to electroosmosis and the pressure gradient, and the electromigration velocity in Eq. 9 gives

$$\frac{\partial C}{\partial t}(1 + \frac{\rho}{n}K_d) = -(\frac{\varepsilon\zeta}{\tau^2\mu}\nabla\phi - \frac{k_h}{n\mu}\nabla p - vzF\frac{\nabla\phi}{\tau^2})\frac{\partial C}{\partial x} + \frac{D}{\tau^2}\frac{\partial^2 C}{\partial x^2} \qquad (11)$$

3. Initial and boundary conditions

Suitable initial and boundary conditions are needed to obtain the numerical solution of Eq.11. Cs^{2+} and Co^{2+} were initially uniformly distributed in the soil column. Since the Cs^{2+} and Co^{2+} concentrations in the solution in the anode reservoir were zero and the solution constantly flowed through the soil column, the initial and boundary conditions were as follows. Also, each electrode was given an electric potential of 100 voltages.

$$C(0,x) = C_0 \quad 0 < x < 20\,(cm)$$

$$C(t,0) = 0$$

$$\frac{\partial C}{\partial x}(t,20) = 0$$

$$\phi(t,0) = 100(V)$$

4. Code development

A numerical code was developed to predict the residual contaminant concentration in the soil in the column after soil remediation by the electrokinetic method. An equation developed on the basis of the governing equation, Eq.11, was rewritten in FORTRAN 77 language. The Galerkin finite element method with a linear basis function, an implicit difference scheme for the space derivative, and a backward difference scheme for the time derivative were used. Also, the Thomas algorithm was used for the matrix calculation. Meanwhile, to avoid any numerical oscillation, the space step (Δx) was selected so that P_e (Pecret number) ≤ 2, and the time step (Δt) so that Cu_e (Courant number) ≤ 1.

III. Soil Remediation Experiment

1. Soil components and measurement of input parameters for simulation

The main components of the TRIGA soil were SiO_2, Al_2O_3 and K_2O. Also, the physical characteristics of the soil, such as the dry bulk density, porosity, and water content are listed in Table 2. The experimental methods are as follows: dry bulk density (ρ_b) is the oven-dried mass of the sample divided by its field volume. Porosity (n) is obtained by 1-(ρ_b / ρ_s). Particle mass density (ρ_s) is the oven-dried mass divided by the volume of the solid particles. Water content (θ) is volume of the water (V_w) divided by the total volume (V_T). In order to obtain an equilibrium sorption coefficient, 6 sets of Cs^{2+} and Co^{2+} solutions with different concentrations were respectively prepared and put into individual plastic bottle containing 10g of soil. The temperature was 25°C, 50 $m\ell$ of Cs^{2+} solution for 10g of soil was used, and the shaking time was 24 hours. Then, the Cs^{2+} concentration remaining in each solution was measured by atomic absorption spectroscopy. The Cs^{2+} quantity sorbed in the soil was calculated. Under the above experimental conditions, the equilibrium sorption coefficient for Cs^{2+} was 8.10 $\ell\,kg^{-1}$, while that for Co^{2+} was 1.25 $\ell\,kg^{-1}$.

2. Remediation using NaCl Solution

The TRIGA soil (<0.063mm) having a low level of radioactivity was artificially contaminated with a 0.01M Co^{2+} solution. Fig. 1 shows the electrokinetic experimental equipment on a laboratory scale. A 0.01 M NaCl solution was used as an electrolyte to heighten the electric field strength. Both reservoirs were filled with the electrolyte solution. The electrolyte solution in the anode continuously moves to the cathode by electroosmosis. Anode reservoir was continuously filled with electrolyte through the upper hole of the reservoir. The quantity of effluent was continuously measured. The Co^{2+} concentration of the effluent was periodically measured with atomic absorption spectroscopy.

Table 2. Simulation Parameters

Parameter	Values
Cell dimensions, L	φ 2.8cm x20 cm
Bulk density, ρ_b	$0.76\,gcm^{-3}$
Faraday's constant, F	$96487\,C\,mol^{-1}$
Potential at anode well, ϕ_1	100.0 V
Potential at cathode well, ϕ_2	0.0 V
Porosity, n	0.61
Tortuosity, τ	1.2
Equilibrium sorption coefficient(K_d ,ml g^{-1})	$8.10(Cs^{2+})$, $1.25(Co^{2+})$
pH of soil	4.3

The experiments for soil remediation were implemented twice. The first experiment was implemented for 6.7 days, and the second for 0.8 days. After the experiment was completed, the kaolin clay in the column was divided into several pieces. The residual Co^{2+} concentration of each piece was then measured with atomic absorption spectroscopy.

Remediation using acetate buffer
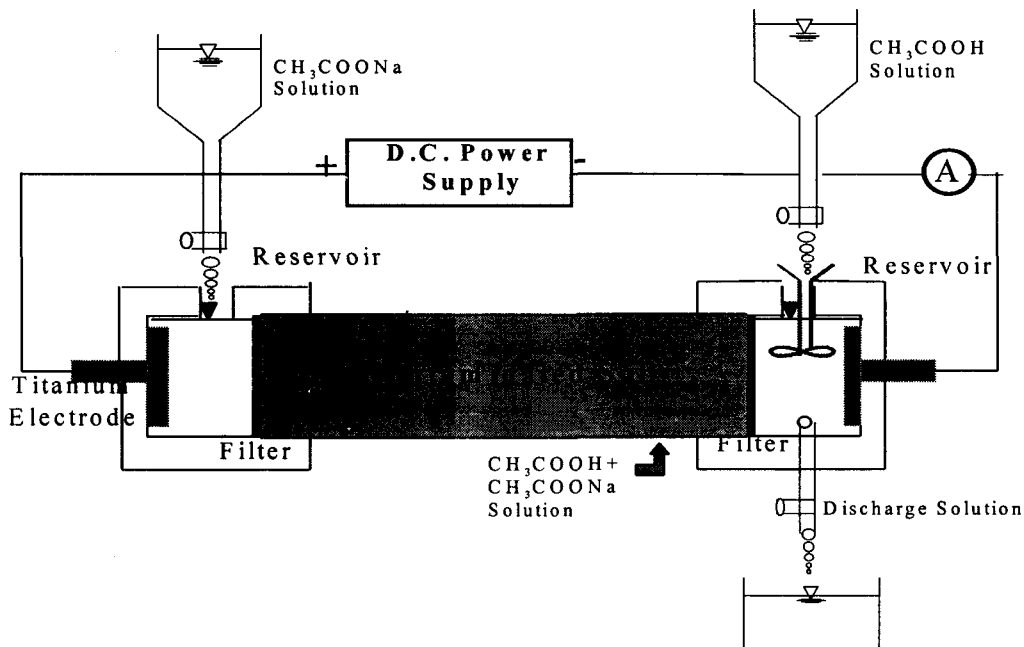
Figure 1. Schematic of cylindrical column for electrokinetic remediation

The TRIGA soil (<0.063mm) having a low level of radioactivity was artificially contaminated with a 0.01M Cs^{2+} and Co^{2+} solution. The option chosen for restraining the pH increase in the soil column was to inject an acetate buffer into the soil column and simultaneously to fill up the cathode reservoir with an acidic solution and the anode reservoir with an alkaline solution. CH_3COONa and CH_3COOH solutions were chosen as the alkaline and acidic solutions respectively. Accordingly, a compound solution of 0.01M of CH_3COONa and CH_3COOH was mixed with the soil to form an acetate buffer solution in the soil column . This buffer solution was chosen keeping in mind the application of the process to an actual in-situ situation, as it is a cheap, biodegradable, and nontoxic reagent. Due to this pH control, a regular electrical potential gradient was expected. The experimental process was run as follows. After the mixed soil including the acetate buffer solution was injected into the column, the experiment was initiated under a constant electric voltage gradient. The volumes of the anode and cathode reservoirs were both 20 ml. 19ml of a 0.01M CH_3COONa solution was initially injected into the anode reservoir, while 13ml of a 0.01M CH_3COOH solution was injected into the cathode reservoir. The pore solution in the column flowed toward the cathode reservoir by electroosmosis. When the cathode reservoir was filled with about 19 ml of the pore solution released from the soil column, the experiment was suspended, then the volume of pore solution in the cathode reservoir was measured exactly. About 3ml of the pore solution was sampled and analyzed for Cs^{2+} and Co^{2+} concentrations by atomic absorption spectroscopy. This process was repeated many times until the experiment was completed. Thereafter, the TRIGA soil was divided into several sections, which were then analyzed for their Cs^{2+} and Co^{2+} concentrations.

The initial solution with $C_{T,Ac} = 0.02M$ was prepared using $C_{HAc} = C_{KAc} = 0.01M$. Since $[H^+]$ and $[OH^-]$ are very small with respect to $[HAc]$ and $[Ac^-]$ in this region, it was assumed that the small amount of [OH^-] added reacted completely with $[HAc]$. Thus, $[HAc]$ decreased, and $[Ac^-]$ increased according to the amount of $[OH^-]$ added per liter. A pH 11.8 equals 0.0063 M of $[OH^-]$. Thus $[HAc]$=0.01–0.0063=0.0027(M) and $[Ac^-]$=0.01 + 0.0063=0.0163 (M). Consequently, the pH in the column was restrained from increasing and predicted to be about 5.48 based on the following calculation:

$$pH = pK_a + \log\frac{[Ac^-]}{[HAc]} = 4.70 + 0.78 = 5.48$$

Meanwhile, the soil contaminated with Cs^{2+} and Co^{2+} was placed in a large bowl and mixed with the acetate buffer solution. Then, the saturated soil was injected into the soil column with tapping. The soil column was continually tapped until almost all the air bubbles had been removed. The soil column was cylindrical in shape with a 2.8cm diameter and 20.0cm length. A titanium electrode, filter, stainless steel net, and reservoir were all established at both ends of the column. The volume of the column was 123 cm^3 and the weight of the TRIGA soil was 110g. The volume of the buffer solution injected into the column was 76 ml. One side of the column was connected with the anode, while the other side was connected with the cathode. The voltage difference between the two electrodes was 100 V and the electric current was about 0.1 mA. A 0.01M CH_3COONa solution was initially injected into the anode reservoir while a 0.01M CH_3COOH solution was injected into the cathode reservoir. Because the CH_3COONa solution in the anode reservoir

continuously moved to the cathode by electroosmosis, this solution was continuously injected through the upper hole of the anode reservoir. The volume of the effluent was measured along, with Cs^{2+} and Co^{2+} concentrations using atomic absorption spectroscopy.

IV. Results and Discussion

When the TRIGA soil was mixed with a NaCl solution to raise the electric field strength for electrokinetic remediation, the pH of the electrolyte in the anode reservoir decreased to a low value immediately, while that in the cathode reservoir increased to a high value. After remediation for 0.8days, the pH at the position 14 cm away from the anode filter in the soil column increased from the initial 4.0 to 11.7, and a precipitate with metal ions started to form at that position. During the experiment period, hydrogen ions evolved at the anode, and hydroxide ions evolved at the cathode. Metal ions migrated from the front part of the column during this period formed a precipitate at the 14 cm position in the column, and continuously accumulated (Fig. 2). As such, the remediation efficiency after 6.7days was significantly decreased due to the small volume and lower concentration of the effluent.
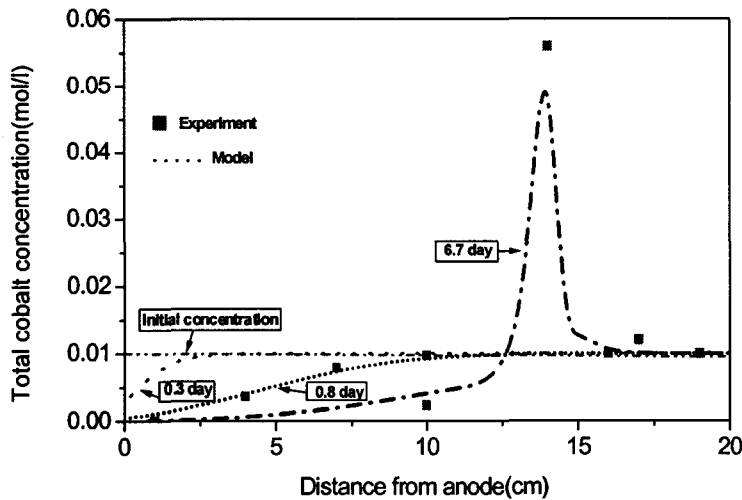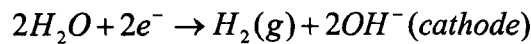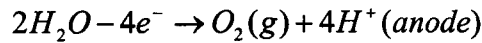
$$2H_2O - 4e^- \rightarrow O_2(g) + 4H^+ (anode)$$
$$2H_2O + 2e^- \rightarrow H_2(g) + 2OH^- (cathode)$$

Figure 2. Distribution of total cobalt concentration in soil colcumn versus time

Fig. 3 shows the effluent pore volume evolved from the soil column relative to time. One pore volume was 75 ml, which was calculated on the basis of the density and porosity of the TRIGA soil. About 4.1 pore volumes of effluent were discharged for 5.9 days. For 4.0days, the average effluent volume rate was 0.6 pore volume/day, while from 4.0 to 5.9 days this increased to 0.9 pore volume/day.

Fig. 4 shows the Cs^{2+} and Co^{2+} concentrations in the effluent from the soil column. The average Cs^{2+} concentration in the effluent was 0.0051M for first 0.5days. Since this concentration was 5.7 times larger than the initial Cs^{2+} concentration in the pore solution, namely 0.0009M, this indicates that many Cs^{2+} ions were transferred

by electromigration rather than electroosmosis during this period. The Cs^{2+} concentration in the effluent abruptly increased to about 0.009M from 0.5days to1.7 days, then decreased to 0.0016M model developed to calculate the residual concentration were fairly similar to the experimented values.
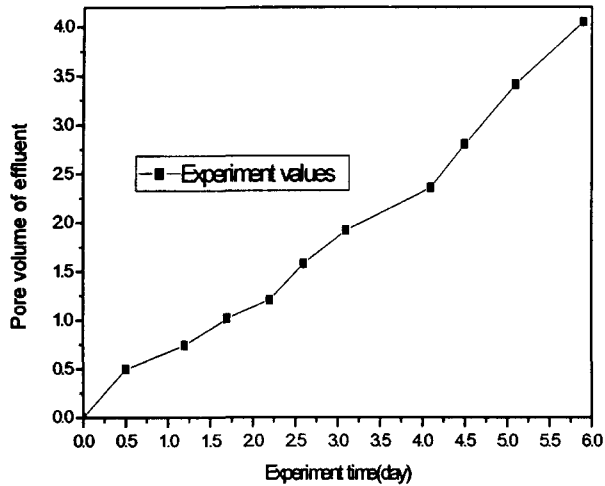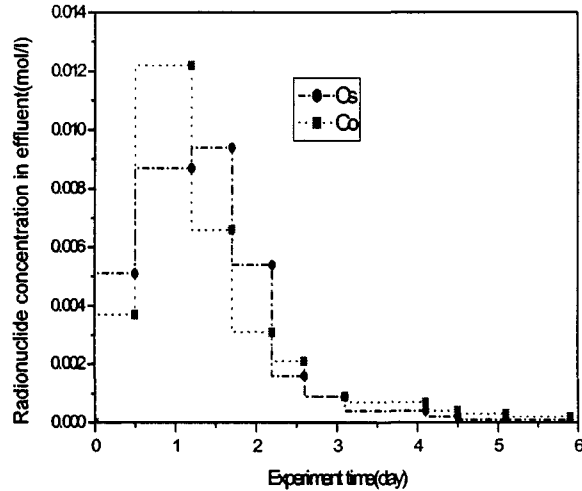

Figure 3. Pore volume of effluent versus time

Figure 4. Cesium and Cobalt concentrations in effluent

rom1.7days to 2.6 days, and further decreased to below 0.001M after 3.1 days. Meanwhile, the average Co^{2+} concentration in the effluent was 0.0037M for first 0.5days, and then increased to 0.0122M from 0.5days to 1.2days. Since this concentration was 3.1 times larger than the initial Co^{2+} concentration in the pore solution, namely 0.0039M, this indicates that many Co^{2+} ions were transferred by electromigration rather than electroosmosis during this period. However, the Co^{2+} concentration of effluent abruptly decreased to about 0.0031M from 1.2 days to 2.2 days, then continuously decreased to below 0.001M after 3.1 days.

The total Cs^{2+} residual concentration in the column versus each remediation time is shown in Fig. 5. Since no precipitate was formed in the column due to the restraint of any pH increase, Fig. 5 showed a high remediation efficiency. Namely, 25.1% of the total amount of Cs^{2+} in the column was removed after remediation for 0.5 day, 44.4% after 1.0 days, 76.1% after 2.0 days, 89.0% after 3.0 days, and 96% after 5.9 days. At the end of the run, the TRIGA soil was drawn out from the soil column and divided into several sections. The concentration of the pore solution was measured using atomic absorption spectroscopy. The values predicted by the developed model were fairly similar to the experimental values.

Meanwhile, Fig. 6 shows the results of the cobalt remediation, which exhibited a similar pattern to that of cesium remediation. Namely, 22.7% of the total amount of Co^{2+} in the column was removed after remediation for 0.5 day, 44.4% after 1.0 days, 71.8 after 2.0 days, 85.2% after 3.0 days, and 94.4% after 5.9 days. The values predicted by the model developed to calculate the residual concentration were fairly similar to the experimented values.
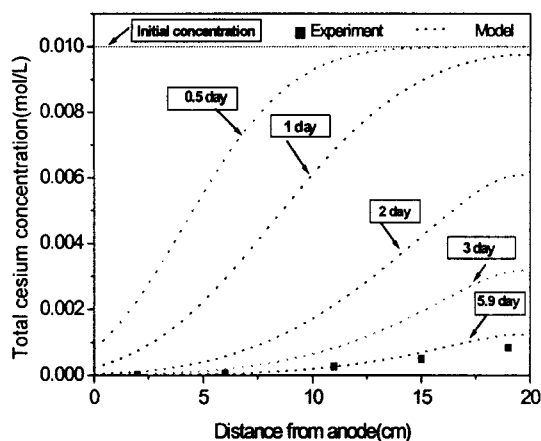
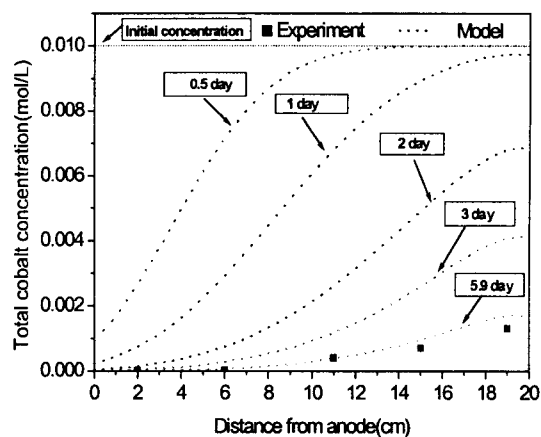Figure 5. Distribution of total cesium concentration in soil column versus time

Figure 6. Distribution of total cobalt concentration in soil column versus time

V. Conclusions

After TRIGA soil with a particle size below 0.063mm was artificially contaminated with a 0.01M Cs^{2+} and Co^{2+} solution, the remediation characteristics of cesium and cobalt using the electrokinetic method were analyzed. An acetate buffer solution was injected into the soil column and a CH_3COOH solution periodically injected into the cathode reservoir to restrain any pH increase. The experimental results indicated no precipitate, such as $Co(OH)_2$, formation in the soil column. Many Cs^{2+} and Co^{2+} ions were transferred by electromigration rather than electroosmosis during initial remediation period. 25.1% of the total amount of Cs^{2+} in the column was removed after remediation for 0.5 day, 44.4% after 1.0 days, 76.1% after 2.0 days, 89.0% after 3.0 days, and 96% after 5.9 days. Also, 22.7% of the total amount of Co^{2+} in the column was removed after remediation for 0.5 day, 44.4% after 1.0 days, 71.8% after 2.0 days, 85.2% after 3.0 days, and 94.4% after 5.9 days. The values predicted by the model developed to calculate the residual concentration after remediation were fairly similar to the experiment values.

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