

Kinetics of electrodialytic extraction of Pb and soil cations from a slurry of contaminated soil fines

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Abstract

The objective of this work was to investigate the kinetics of Pb-removal from soil fines during electrodialytic remediation in suspension, and study the simultaneous dissolution of common soil cations (Al, Ca, Fe, Mg, Mn, Na and K). This was done to evaluate the possibilities within control of the remediation process to leave a final product suitable for reuse. The Pb-remediation process could be divided into four phases: (1) a “lag-phase”, (2) a period with a high removal rate (7.4 mg/day in average at 40 mA), (3) a period with a low removal rate, and (4) a period where no further Pb-removal was obtained. During the first phase, dissolution of carbonates was the prevailing process, resulting in a corresponding loss of soil mass. During this phase, the investigated ions accounted for the major current transfer, while, as remediation proceeded, hydrogen ions increasingly dominated the transfer. During phase (3) the high conductivity and low voltage suggested that removal may be accelerated by increasing the current density. Overall, 97% of the Pb could be extracted, reducing the final Pb-concentration to 25 mg/kg. The order of removal rates was: Ca > Pb > Mn > Mg > K > (Al and Fe).

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1. Introduction

Sludge of heavy metal contaminated soil fines is the main process residue from soil washing, which severely limits the success of the process for remediation of heavy metal contaminated soil. Electrodialytic soil remediation (EDR) is an electrokinetic remediation (EKR) method, where ion-exchange membranes are applied as barriers between soil and electrolytes. It was recently shown that during EDR of soil fines in suspension, Pb is easily dissolved by the acidification resulting from water splitting and transferred to the catholyte [1]. Potential applications of the remediated soil fines include: ceramics (bricks and tiles), lightweight-expanded aggregates, cement, concrete, soil amendment, and landfill liners. Besides the content and leachability of heavy metals, the applicability of the treated soil fines depends on their final characteristics including content of salts, nutrients and minerals: for production of bricks and roof tiles iron-oxides are unwanted [2]; in lightweight-expanded aggregates

the expanding qualities depend on the grain size and the composition of the clay minerals; in cement production, chloride is problematic [2]; in production of concrete, water-soluble species are in general unwanted [2]; and for application as landfill liners or as soil amendment, the acidity of the treated soil fines is a limitation.

During the EDR process, the mineralogy of the soil fines is subject to alterations, and attention needs to be paid towards the quality of the treated product prior to application. Dissolution of natural soil constituents during EDR/EKR was demonstrated in several studies [3–5]. Ca dissolution was observed to precede contaminant removal and coincide with the pH-shift in the soil [3,4]. Also Fe, Mg and Mn were removed from soil as a response to EKR [4]. Here, removal of Mg and Mn was directly related to the pH-decrease, while no relation between pH and Fe-removal was observed. Mineral dissolution was further demonstrated by West et al. [5], who observed transfer of Na, Mg, Ca, K, Al, Fe, and Si ions from kaolinite into the electrolytes during EKR of spiked kaolinite.

Studies on the influence of direct current on clay minerals were made in the mid 1900s. It was shown that extensive hydration of common minerals (olivine, augite, hornblende, pargasite,

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biotite, chlorite, muscovite and feldspar) took place as a response to direct current [6]. The hydration was accompanied by dissolution and transport of K, Na, Ca, Fe, Mg, Al and Si. The authors held that the drastic changes imply a partial or complete destruction of the geometry of the original minerals. Evidence of mineralogical changes in a thin surface-film covering the original mineral was given, however the XRD technique was not sophisticated enough to make ultimate conclusions at that time. A great variance in stability among minerals towards electroanalysis was demonstrated in a study, where extensive dissolution of saponite clay and removal of Si was demonstrated, while only little Si was removed from nontronite clay [7]. Similarly biotite and jeffersite were almost completely decomposed while muscovite and phlogopite only gave of small quantities of cations [8]. Interestingly, the XRD-patterns of the biotite and jeffersite did not change during treatment. A recent study showed how montmorillonite exhibited color-changes (grey to green), shrinkage cracks, water loss and an increased Fe(II)-content as a response to the direct current and the acidic front. These reactions concurrently pointed towards reduction of octahedral Fe(III) to Fe(II) within the lattice of the clay mineral [9].

In the present work, the dissolution and removal of a contaminating heavy metal (Pb) was investigated and compared with the removal of major soil cations (Fe, Al, Mg, Na, Mn, Ca, K). The removal rates were studied, and the processes during EDR are discussed.

2. Materials and methods

2.1. Analytical procedures

Pb, Fe, Al, Mg, Na, Mn, Ca, and K were analyzed by flame AAS. Samples with Pb-concentrations below the detection limit (1 mg/L) were measured by graphite furnace AAS. Prior to analysis of soil samples, 1.00 g soil fines were digested in autoclave with 20.00 mL 1:1 HNO₃ for 30 min at 120 °C and 200 kPa according to the Danish standard method DS259 [10], and filtered through a 0.45 µm filter by vacuum. Validation of AAS results was obtained by measurement of liquid reference samples. The carbonate content was determined by the volumetric calcimeter method described in [11]. In calculations it was assumed that all carbonate is present as calcium carbonate. Organic matter was determined by loss of ignition at 550 °C for 1 h. CEC was measured with a method comparable to the acid-NaCl method described in EPA Standard Method 9080. pH and conductivity were measured by a Radiometer Analytical electrode. All measurements were made in triplicate.

2.2. Soil

An industrially contaminated Danish soil of unknown origin was used as experimental soil. The soil fines were obtained by wet-sieving of the original soil with distilled water through a 0.063 mm sieve. A concentrated slurry of fines was obtained by centrifugation at 3000 rpm for 10 min, and decantation of the supernatant. The soil fines were kept in slurry and stored at 5 °C in access of atmospheric air. The mineralogy of the original soil

Table 1
Characteristics of the soil fines

Pb (mg/kg)	673 ± 101
Mn (mg/kg)	542 ± 49
Ca (g/kg)	64.7 ± 4.7
Mg (g/kg)	4.3 ± 0.7
Fe (g/kg)	23.2 ± 4.6
Al (g/kg)	12.5 ± 3.1
K (g/kg)	4.0 ± 0.9
Na (mg/kg)	410 ± 50
CaCO ₃ (%)	17.3 ± 0.1
Organic matter (%)	7.8 ± 0.1
CEC (mequiv./100 g)	14.1 ± 0.3

(comprising both the coarse and the fine fractions) was investigated by XRD in [12] (soil 10). It was found to contain the following mineral phases: quartz, K-feldspar, plagioclases, calcite, kaolinite, illite, smectite and mixed elementary layers of the three clay minerals. Initial soil characteristics and content of soil cations are listed in Table 1.

2.3. Remediation experiments

Electrodialysis experiments were made in cylindrical Plexiglas®-cells with three compartments. Compartment II, which contained the soil-slurry, was 10 cm long and 8 cm in inner diameter. The slurry was kept in suspension by constant stirring (1600 rpm) with plastic-flaps attached to a glass-stick and connected to an overhead stirrer (RW11 basic from IKA). The anolyte was separated from the soil specimen by an anion-exchange membrane, and the catholyte was separated from the soil specimen by a cation-exchange membrane. Both membranes were obtained from Ionics® (types AR204SZRA and CR67 HVY HMR427). Fig. 1 shows a schematic drawing of the setup. Electrolytes were circulated by mechanical pumps (Totton Pumps Class E BS5000 Pt 11) between electrolyte compartments and glass bottles. Platinum coated electrodes from Permascand® were used as working electrodes, and the power supply was a Hewlett Packard® E3612A. The catholyte and anolyte initially consisted of 0.01 M NaNO₃ (500 and 300 mL, respectively) adjusted to pH 2 with HNO₃. In previous experiments the build-up of an osmotic pressure difference between

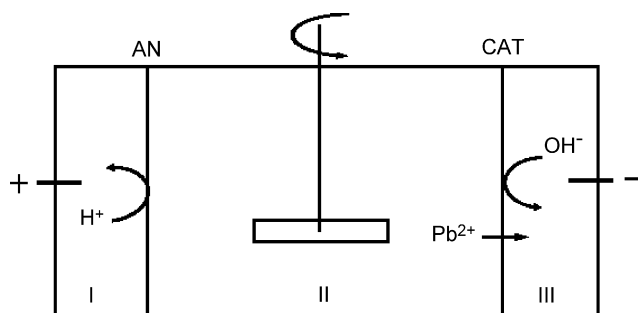


Fig. 1. Schematic view of a cell used for experimental EDR remediation of soil fines in suspension. AN, anion-exchange membrane; CAT, cation-exchange membrane. I, anolyte compartment; II, compartment containing slurry of soil fines; III, catholyte compartment.

Table 2
Experimental results

Experiment	Pb mass balance (%)	Soil mass balance (%)	Final Pb (mg/kg)	% Pb removed from soil	% Pb in liquid II	Final pH
1	108	87	798	5	0	6.9
2	102	84	224	73	33	1.5
3	103	82	110	87	1	1.5
4	86	81	44	94	2	1.1
5	73	74	23	97	6	1.0
6	93	78	27	97	1	1.4

the two compartments was observed to result in an extensive water transfer from compartment I to II. This water transfer was avoided by decreasing the amount of liquid to be circulated in the anolyte from 500 mL, used in previous works [1] to 300 mL, whereby the hydraulic pressure was decreased and overflow avoided.

Current, voltage, pH in all compartments, and conductivity in compartment II were measured approximately once every 24 h. pH in the electrolytes was kept between 1 and 2 by manual addition of HNO₃ (7 M) or NaOH (6 M). The six experiments were identical apart from the duration which was 188, 330, 503, 671, 838, and 930 h, respectively. The liquid to solid ratio (L/S) was 4.3 (87 g soil and 375 mL distilled water), and the current density was 0.8 mA/cm² (40 mA).

After each experiment, membranes were cleaned overnight in 1 M HNO₃, and electrodes were cleaned overnight in 5 M HNO₃. Volumes of the cleaning acids, the electrolytes, and the solution in the middle compartment were measured followed by analysis of the cation-concentrations by AAS. The remaining soil mass was decided and the cation concentrations in the soil were measured by AAS after digestion according to DS259 as described above. The mass balances for Pb and soil cations were calculated as the mass of the individual elements found in the whole system after remediation (in soil, soil solution, electrolytes, membranes, and at electrodes) in percent of the amount found in the soil prior to remediation (concentration times initial amount of soil). The current efficiency was calculated for each of the elements of interest. The current efficiency is defined as the fraction of the current passing through an electrolytic cell that accomplishes a specific chemical reaction (in this case transfer of a specific element (cation) from the soil fines into the catholyte). The current efficiency for specie *x* was calculated as follows:

$$\varepsilon_x = \frac{Q_x}{Q_{\text{tot}}} = \frac{M_x z_x / MW_x}{It / F}$$

where Q_x is the amount of current (moles) which was transferred by specie *x*, and Q_{tot} is the total amount of current passed through the cell during remediation. M_x is the mass of the specie *x*, which was transferred out of the soil fines, z_x is the valence of the specie *x*, MW_x is the molar weight of the specie *x*. I is the current (in A) passed through the cell, t is the experimental time (in seconds), and F is the Faraday constant. The calculation was based on the assumption that the individual elements were transferred with the following valences: Pb²⁺, Mn⁴⁺, Ca²⁺, Mg²⁺, Fe³⁺, Al³⁺, K⁺, Na⁺.

3. Results and discussion

3.1. Pb-removal

Main results of the six experiments are summarized in Table 2. Pb mass balances between 73 and 108% were obtained. The final amount of soil was reduced with 13–26% compared to the initial amount. This general reduction is likely to reflect a partial dissolution of soil constituents as discussed in Section 3.3. The final Pb-concentration in the soil fines was between 798 and 23 mg/kg with a clear reduction as remediation time increased, and with 97% removal in the two experiments of the longest duration (calculated as the fraction of the Pb found elsewhere than the soil after remediation). In experiment 2, 33% of the Pb appeared in the soil solution in compartment II, while in all other experiments the amount of Pb in dissolution in II was minor. The final pH of the soil solution was between 1 and 2 for all experiments except the one with the shortest duration, where pH had only decreased slightly from the initial value (7.5) to 6.9.

The Pb-removal from the soil is illustrated as a function of time in Fig. 2 (one point for each experiment) together with the concentration build-up in the cathode section (catholyte, cathode and cation-exchange membrane), the anode section (anolyte, anode and anion-exchange membrane), as well as the Pb dissolved in the solution in the middle compartment (II). In all experiments >90% of the Pb in the cathode section was precipitated at the cathode itself.

The highest removal rate was obtained between 188 and 330 h. Here, the average rate was 7.4 mg/day. In order to

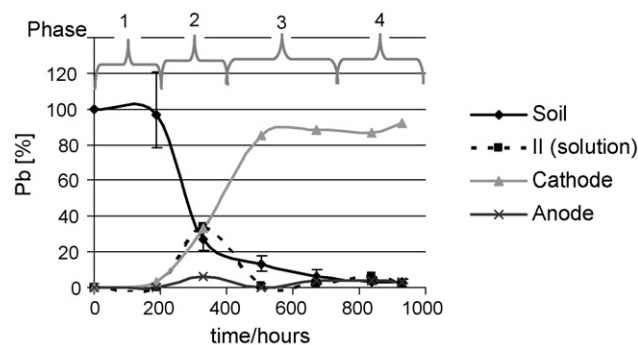


Fig. 2. Removal of Pb from soil, dissolution in solution in soil solution (II) and concentration in anode and cathode sections. Removal proceeds in four phases: (1) lag-phase; (2) high removal rate; (3) low removal rate; (4) removal stopped.

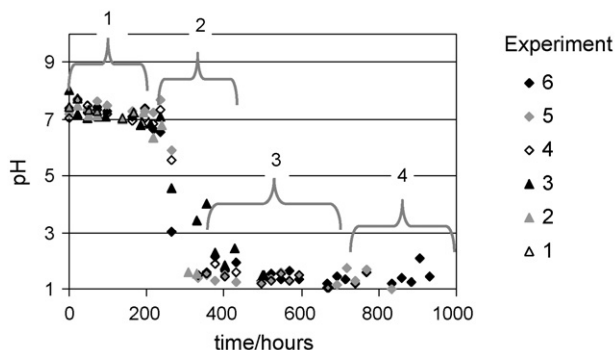


Fig. 3. pH-development in the soil-slurry (compartment II) during remediation.

determine the maximum obtainable rate, more experiments in this interval would be necessary, and further optimization options should be investigated. This work however clearly illustrates how the removal undergoes four phases (1) a “lag-phase”, where removal is substantially absent, (2) a period with a high removal rate involving dissolution of Pb in the soil solution, (3) a period with a low removal rate, where the dissolved Pb is removed from solution, and (4) a period where no further Pb-removal is obtained as the treatment proceeds.

3.2. pH, conductivity and voltage

The pH-development in the soil solution during remediation is illustrated in Fig. 3. pH was more or less constant during the first 200–240 h, followed by a sharp decline to 1–2, where pH stabilized after 360–400 h. This pH-decline is consistent with previous observations [1], and is believed to occur due to water splitting at the surface of the anion-exchange membrane induced by insufficient amounts of anions available for current transfer in the solution [1]. The sharp pH-decrease coincided with the maximum rate of Pb-removal, and confirms that acidification is the foundation of unenhanced EDR of HM-containing materials [13]. This result clearly shows how the acidification is responsible for desorption and dissolution of the Pb, thus the acidification should be regarded as a prerequisite for the success of the remediation method. The response to the acidification is made visible by the fact that a large fraction of Pb was found in solution in II by the end of experiment 2 (Fig. 2). This result illustrates, how the sudden acidification results in a prompt release of Pb (and other ions) in excess of the charge transfer. However, after a short while (experiment 3), all the released Pb had been removed from the solution by the current, and during phase (3) of the remediation, Pb was removed from solution as it was released from the soil, which indicates a desorption limited removal in this phase. In addition, the attainment of the low pH plateau coincided with the change from the high to the low removal rate, suggesting that the removal rate decreases due to an overflow with H^+ -ions, which compete successfully for the current transfer. The last phase (4), where no removal took place, was not related to any pH-changes, but rather occurred because most of the anthropogenic and available Pb at that point had been removed. This assumption is verified by the fact that also no changes in con-

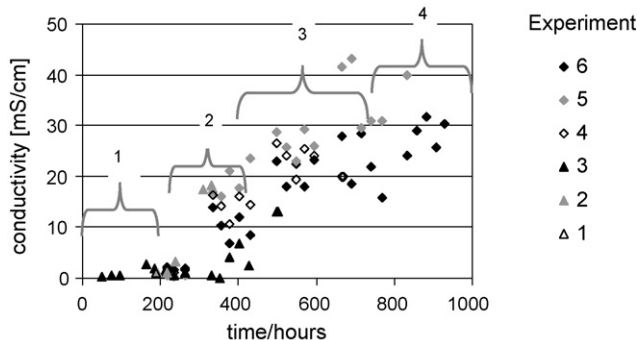


Fig. 4. Conductivity of soil-slurry during remediation.

ductivity (Fig. 4) and voltage (Fig. 5) were observed between phases (3) and (4).

Developments in the conductivity of the soil-slurry are illustrated in Fig. 4. During remediation, the conductivity increased as a prompt response to the pH-decrease (phase 2), confirming how the Pb-removal declined due to preferential transfer of hydrogen ions in the acid environment. The voltage between the working electrodes decreased (Fig. 5) as a response to the acidification and the resulting increased conductivity. In the period prior to acidification several incidents of high voltage (>80 V) occurred. These incidents coincided with observations of high pH in the catholyte (data not shown) caused by OH^- production by the electrode-process. As soon as pH in the catholyte was regulated down by addition of nitric acid, voltage decreased again.

These incidents of high voltage are likely to have occurred due to precipitation of hydroxides within the cation-exchange-membrane, and suggest that faster removal and/or lower energy consumption could have been obtained through pH-static control of the catholyte. By the end of the acidification, the simultaneous conductivity increase and voltage-decrease suggest that a higher current might have been forced on the system during phase (3) in order to maintain a high removal rate. During phase (1) however, voltage in general was between 20 and 40 which is above the previously [1] recommended value (maximum 20), suggesting that some water splitting at the cation-exchange membrane may have taken place and prolonged this phase unnecessarily. A lower current density might have accelerated the process during this phase.

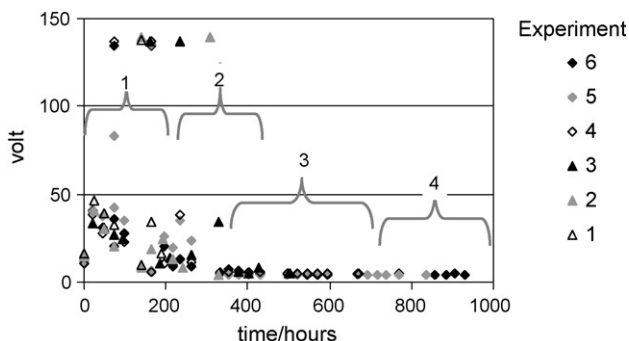


Fig. 5. Voltage between working electrodes during remediation.

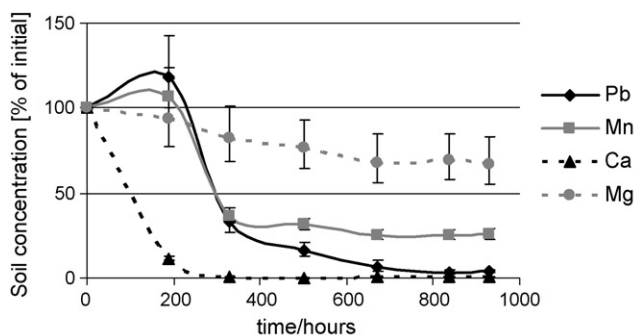


Fig. 6. Dissolution of Pb, Mn, Ca and Mg from the soil.

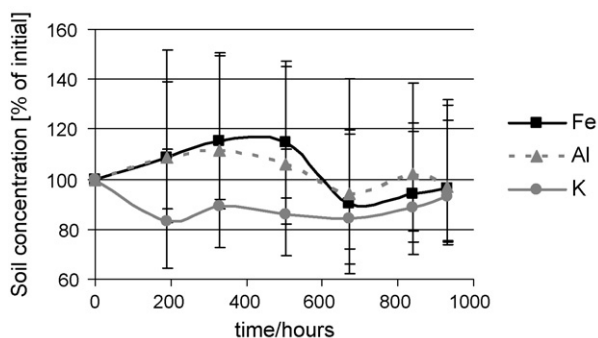


Fig. 7. Dissolution of Al, Fe and K.

3.3. Extraction of soil cations

Figs. 6–8 illustrate the influence of the electro-dialytic treatment on the soil's content of common soil cations. The concentration of each element is given in percent of its initial concentration, thus elements which do not dissolve as fast as the average soil will obtain a concentration above 100%. In general it should be noted that due to the extraction procedure, only the fraction of the elements which is not bound within the silica matrix is included in the discussion. In accordance with [5,6], we observed transfer of Na, Mg, Ca, K, Al, Fe from the soil into the electrolytes. In Fig. 6 the extraction of the cations most affected by the electro-dialytic treatment (Mn, Ca and Mg) is illustrated in relation to the extraction of Pb.

Ca was completely extracted during phase (1), thus prior to Pb. The early extraction of Ca is in accordance with results of [3,4], and shows that the prevailing process during phase (1) is

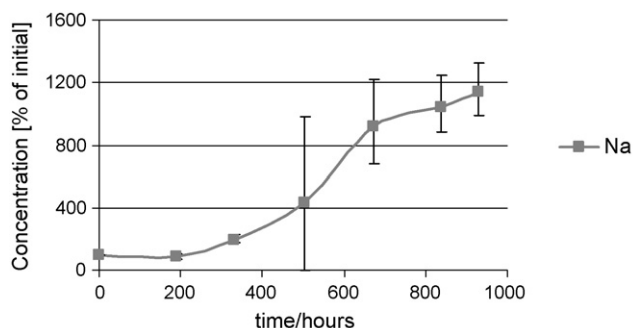


Fig. 8. Intrusion of Na into soil.

elimination of the soils buffer-capacity. The dissolution of primarily calcium carbonates followed by removal of Ca explains the observed soil mass reduction (Table 2) well: (a) the reduction in soil mass in the experiment with the shortest duration and with incomplete Ca-removal was 13%, which is below the content of calcium carbonate (17%); (b) the average reduction in the remainder experiments was 20% which is just above the amount of calcium carbonate in the soil, confirming that calcium carbonate is the primary soil-constituent undergoing dissolution; (c) the total amount of extracted Ca was equivalent to a calcium carbonate content of 16%, confirming that calcium carbonate is the major carbonate source, but that other less abundant carbonates are present as well. Of those, the most common is dolomite $\text{CaMg}(\text{CO}_3)_2$, of which dissolution was confirmed by a slight decrease in the Mg-concentration of the soil fines during phase (1), where, in contrast, the concentration of Pb and Mn increased.

During the following phases (2 and 3) the slow dissolution of Mg continued, while it ceased in phase (4) after the Mg concentration had been reduced with 25% (50% Mg-extraction). The fact that the Mg-dissolution was not related to the pH-shift is in contrast to the findings of [4], and suggests that a main fraction of the Mg in this soil was bound in soil-minerals, which were less affected by the EDR process.

The increased concentrations of Pb and Mn observed after 188 h of EDR appeared due to the described preferential extraction of carbonates and the resulting reduced soil-volume, which caused unaffected elements to concentrate in the soil-phase. Mn-extraction further resembled Pb-extraction in that it occurred at a high rate during phase (2), at a low rate during phase (3) and ceased in phase (4). The relation between the pH-shift and Mn-release is consistent with the results of [4]. The Mn-concentration was however only reduced with 75% (80% Mn-extraction) before extraction ceased, suggesting that as for Mg, a fraction was bound in stable soil-minerals, which were less affected by the EDR process.

Extraction of Fe, Al and K is illustrated in Fig. 7 (note the different y-scale). The large variation on analytical results primarily reflects a large variation on the analysis of the initial content. Cautious interpretations may however still be made. It seems that the concentration of Fe and Al increased throughout phases (1), (2) and part of phase (3). This shows how Fe- and Al-minerals were relatively unaffected by the EDR process during those phases, and is in accordance with the previous observation that Fe-removal was not related to the sharp pH-decrease [4]. At the low pH-level prevailing in phases (3) and (4), they, however, began to dissolve, and Fe- and Al-oxides probably acted as buffers at this low pH-level. Dissolution of particularly Al is undesirable during EDR due to its toxicity. Therefore, according to these results, it is recommended to terminate remediation as soon as the Pb-extraction ceases. K was dissolved during phase (1) to a larger extent than Fe and Al, but dissolution ceased after phase (2). Because of the reduced soil mass, the concentration of all three elements had only decreased slightly compared to the initial concentration by the end of the experimental remediation. This, however, does not mean that the mineral composition was unaffected, as the final amount of extracted Fe, Al and K constituted 25–30%.

Table 3
Current efficiency (%)

Experiment	Element								Total
	Pb	Al	Ca	Fe	K	Mg	Mn	Na	
1	0.10	25	900	23	9	20	1	1	980
2	0.59	17	567	7	5	19	5	−2	618
3	0.67	22	373	9	4	15	4	−5	422
4	0.45	29	279	30	3	14	3	−10	348
5	0.30	24	224	27	2	12	2	−8	283
6	0.36	22	202	19	2	11	2	−9	248

Finally, in Fig. 8, the behavior of Na during EDR is illustrated. This graph shows how the content of Na increased continuously during phases (2), (3) and (4). Sodium nitrate (0.01 M) constituted the initial electrolyte solutions, and clear evidence of intrusion of Na into the soil solution and adsorption onto the soil is given here. This intrusion appeared despite the separation of anolyte and suspension by an anion-exchange membrane and could be due to transfer of Na as co-ions over this membrane and/or diffusion against the current-direction due to the concentration-difference over the cation-exchange membrane, which separated the slurry from the catholyte. The final amount of Na adsorbed to the soil exceeds the initial amount of Na in the electrolytes, proving that some transfer of Na from the anolyte, which was adjusted to 1–2 regularly by addition NaOH, occurred. In addition, the soil may have been enriched in NO_3^- due to regulation of pH in the catholyte with HNO_3 . This observation illustrates how the ion-exchange membranes are not to be conceived as perfect barriers, and demonstrates how the electrolytes should be carefully chosen in order to fulfill any requirements for the soil fine composition prior to subsequent application.

The overall order of removal rate found was: $\text{Ca} > \text{Pb} > \text{Mn} > \text{Mg} > \text{K} > (\text{Al} \text{ and } \text{Fe})$. This is consistent with the order of extraction observed for the clays saponite and nontronite (both smectites), while for attapulgite (palygorskite, not a smectite), Mg was extracted at a higher rate than Ca [7]. Concerning the possibility of control of the remediation process to leave a final product suitable for reuse, our results show that Ca extraction and carbonate dissolution is inevitable during the remediation process. On the other hand, the intrusion of Na shows how it is possible to control the concentration of various ions in the soil by appropriate composition of the electrolytes and choice of acids and bases for pH-regulation. Furthermore, it is visible from the results that only a fraction of the soil's Mn, Mg, K, Fe and Al minerals are dissolved during the process, and that the extent of dissolution of Mg, K, Fe and Al may be limited by adequate process control. Based on the reuse options mentioned in the introduction, it may be possible to decrease the amount of iron-oxides for production application in bricks and roof tiles by continuation of the EDR process beyond the point where Pb-removal has ceased; for cement and concrete production chloride and other water-soluble species are inevitably removed during the process if it is continued beyond phase 2; for application in landfill liners or as soil amendment, the acidity of the treated soil fines will have to be regulated after treatment as the acid-

ification is necessary for the success for the process and thus cannot be limited; while for use in lightweight-expanded aggregates further studies of the mineralogy (e.g. by XRD techniques) is necessary to tell if the necessary expanding clay minerals are preserved during the process.

3.4. Current efficiency

The current efficiency is given in Table 3 for all investigated elements. In experiment 1 almost all the current was accounted for by calcium transport, which again shows how dissolution of carbonates and removal of Ca was the prevailing process during the first stage of EDR. As the remediation proceeded, less and less current was transferred by the investigated soil cations due to the preferential transfer of the produced hydrogen ions during the acidification in phase 2. Although concentrations of Ca, Pb, and Mn were reduced significantly, only Ca-transfer constituted a significant fraction of the current transfer due to the relatively small initial concentrations of Pb and Mn. Conversely, Fe and Al ions, which were not particularly reduced in concentration, constitute a substantial fraction of the current transfer due to their initially high concentrations.

4. Conclusions

Several potential applications of soil fines after electro-dialytic remediation in suspension exist depending on the characteristics of the remediated product. The process of electro-dialytic remediation of Pb-contaminated soil fines can be divided into four phases: in phase (1) the soil buffer capacity is eliminated by the production of hydrogen ions at the surface of the anion-exchange membrane, where water-splitting takes place. During this phase soil-carbonates are dissolved, resulting in a complete extraction of Ca and partial extraction of Mg and K. The carbonate extraction results in a corresponding loss of soil mass, and imply a concentration of elements unaffected by EDR during this phase, including Pb. In this phase the major current transfer can be accounted for by Ca. During phase (2) a sharp pH-decrease of the soil-slurry takes place along with increased conductivity. The pH-decrease results in dissolution of Pb, and although Pb-removal occurs at the highest rate during this phase, a significant fraction of the Pb remains dissolved in the soil solution. Along with Pb, also Mn is extracted. Mg is continuously being extracted during this phase, however at a much lower rate than that of Pb and Mn. In phase (3) pH stabilizes at 1–2, while

the conductivity continues to increase and the voltage between working electrodes decreases. During this phase Pb is removed from solution and extracted at a lower rate, which is likely to be limited by the desorption process, and simultaneously with low-rate extraction of Mn and Mg. Furthermore, Fe and Al-oxides start to act as buffers, resulting in some extraction of these elements as well. In phase (4) extraction of Pb ceased, probably due to the strong bonding of the remaining non-anthropogenic Pb, and the primary transport is that of hydrogen ions complemented by a continuing slow dissolution of Fe and Al-oxides, which act as a buffer at the low pH. Based on the results it is recommended to terminate remediation as soon as Pb-extraction ceases to limit the dissolution of Fe and Al-minerals. Due to intrusion from the electrolytes, the soil content of Na is continuously increasing during remediation, and a careful choice of electrolytes in order to meet requirements by the succeeding application of soil fines is necessary. Ninety-seven percent of the Pb could be extracted, reducing the final Pb-concentration to 25 mg/kg. The overall order of removal rate found was: Ca > Pb > Mn > Mg > K > (Al and Fe). In order to establish a complete evaluation of the product for succeeding applications, this investigation should be complemented by investigations of the fate of phosphate, nitrate, chloride and organic matter as well as the mineralogical condition of the fines after remediation.

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