Electrodialytic remediation of suspended soil – Comparison of two different soil fractions

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ABSTRACT

Electrodialytic remediation (EDR) can be used for removal of heavy metals from suspended soil, which allows for the soil remediation to be a continuous process. The present paper focused on the processing parameters for remediation of a soil polluted with Cu and As from wood preservation. Six electrodialytic treatments lasting from 5 to 22 days with different liquid to solid ratio (L/S) and current intensity were conducted. Among treatments, the highest removal was obtained from the soil fines with 5 mA current at L/S 3.5 after 22 days where 96% of Cu and 64% of As were removed. Comparing the removal from the original soil and the soil fines in experiments with identical charge transportation, higher removal efficiency was observed from the soil fines. Constant current with 5 mA could be maintained at L/S 3.5 for the soil fines while not for the original soil. Doubling current to 10 mA could not be maintained for the soil fines either, and doubling L/S to 7 at 5 mA entailed a very fast acidification which impeded the removal. The results showed that a very delicate balancing of current density and L/S must be maintained to obtain the most efficient removal.

1. Introduction

Electrodialytic soil remediation (EDR) is one of a group of electrochemically based soil remediation methods whose purpose is to remove heavy metals from polluted soil. EDR was originally applied to soil that was moist and consolidated in attempts at in situ treatment [1]. A faster and continuous process was then developed, which can be used ex situ [2–4]: the soil is suspended in a solution (most often water) during such treatment. The overall idea is to combine the method with soil washing and develop a continuous process for heavy metal removal from the fine fraction. Larger debris or soil particles are separated out by the washing procedure, leaving only a highly contaminated sludge for EDR. The soil portion containing sand needs only initial rinsing treatment because contaminants do not strongly adhere to the sand particles. While for the fine fractions like silt and clay, need more extensive remedial treatment because contaminants are easily adsorbed by this fine-grained fraction [5]. The adsorption is either specific or nonspecific, or both of them, which depends on the clay mineralogy and the composition of soil organic matter.

In a reported study, remediation of soil fines (<63 μm) in suspension in distilled water was shown to be efficient for the removal of Pb, and a maximum of 96% of Pb was removed [2]. The method was also used for the remediation of soil polluted by the wood preservation industry, but the pollutants in this soil were not concentrated in the fine fraction as had been expected. As the pollutants were also found in the larger soil particles, soil washing as pretreatment was not possible [6]. In this case the major soil body (<4 mm) was treated in suspension using EDR.

In the present paper, electrodialytic remediation of the original soil (<2 mm) is compared to the remediation of the soil fines (<63 μm). No such comparison appears to have been made. The hypothesis is that remediation of suspended soil fines is more efficient than remediation of suspended original soil, not only because a large fraction of the material is left for simpler and cheaper soil washing, or because the material is kept suspended, thereby reducing the concentration polarisation and resistance [7], but also due to the higher conductivity of the soil fines, which is expected to allow a higher current density and thus faster remediation. Further, as the influence of L/S and current are considered to be important basic parameters, this work focuses on elucidating their role in the remediation process.

2. Materials and methods

2.1. Experimental soil

The soil was sampled from the top layer on an industrial site in Denmark, which had been highly polluted by a wood preservation plant. This investigation only considered Cu and As since

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the Cr is well below the limiting value. The soil was air dried and sieved, and only the particles with size <2 mm were used. The term "original soil" was used for <2 mm soil particles. The "soil fines" were obtained by wet-sieving the original soil with distilled water through a 0.063 mm sieve. Concentrated dry soil fines were obtained by evaporating water on a heating plate under non-boiling condition.

2.2. Analysis of soil characteristics

The original soil and the soil fines were analysed for the following parameters. The concentrations of Cu and As were determined after pretreatment of the soil according to Danish Standard 259, where 1.0 g of dry soil and 20.0 mL (1:1) HNO_3 were heated at 200 kPa (120 °C) for 30 min. The liquid was separated from the solid particles by vacuum through a 0.45 mm filter and diluted to 100 mL. The concentrations of Cu and As were measured with AAS and ICP, respectively. Soil pH was measured in two ways: by suspending 10.0 g dry soil in either 25 mL 1.0 M KCl or 25 mL distilled water. After 1 h of agitation, pH was measured using a Radiometer pH electrode. The content of organic matter was found as a loss of ignition after 1 h at 550 °C. Carbonate content was determined volumetrically by the Scheibler method, which reacted 3 g of soil with 20 mL of 10% HCl. The amount was calculated and assumed that all carbonate was present as calcium carbonate. SEM-EDX analysis was performed on the original soil and the soil fines. The accelerating voltage of the SEM was 15 kV with a large field detector (and X-ray cone). Different areas of the sample were investigated by SEM and the element distribution was examined by element mapping using EDX.

2.3. Desorption of heavy metals as a function of pH

To examine the pH dependent desorption of Cu and As from the original soil and the soil fines, the following procedure was used: 5.0 g dry soil (dried at 105 °C for 24 h) and 25 mL HNO_3 in various concentrations (from 0.01 M to 0.9 M) were suspended for 48 h. The suspensions were filtered (0.45 mm) and the Cu and As concentrations were measured in the liquid phase with AAS and ICP respectively. Extractions in distilled water were made as a reference.

2.4. Sequential extraction of heavy metals

Sequential extraction was performed according to the method described in the Standards, Measurements and Testing Program of the European Union including (1) carbonate and exchangeable, (2) reducible, (3) oxidiseable, and (4) residual fractions, respectively, 0.5 g of dry and crushed soil was treated in four steps as follows: (1) extraction with 20.0 mL of 0.11 M acetic acid (pH 3) for 16 h, (2) extraction with 20.0 mL of 0.1 M $NH_2OH\cdot HCl$ (pH 2) for 16 h, (3) extraction with 5.0 mL of 8.8 M H_2O_2 for 1 h and heating to 85 °C for 1 h with a lid followed by evaporation of the liquid at 85 °C until it had been reduced to less than 1 mL by removal of the lid. The addition of 5.0 mL of 8.8 M H_2O_2 was repeated, followed by resumed heating to 85 °C for 1 h and removal of the lid for evaporation until almost dry. After cooling, 25.0 mL of 1 M NH_4OOCCH_3 (pH 2) was added, and extraction lasted for 16 h, and (4) digestion according to DS 259 with 20.0 mL (1:1) HNO_3 under the condition of 200 kPa and 120 °C was made for identification of the residual fraction. Between each step the sample was centrifuged at 3000 rpm for 15 min, and the supernatant was decanted and stored for AAS analysis. Before addition of each new reagent, the sample was washed for 15 min with 10.0 mL of distilled water and centrifuged at 3000 rpm for 15 min, and the supernatant was then decanted. All extractions
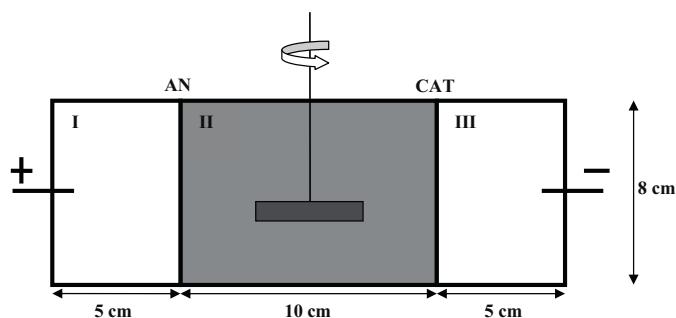

Fig. 1. Principle of electrodialytic remediation of suspended soil. (AN = anion exchange membrane, CAT = cation exchange membrane.)

were performed at room temperature, and samples in each step were taken in triplicate.

2.5. Experimental setup and experiments conducted

The electrodialytic experiments were conducted in cylindrical cells, as shown in Fig. 1. The cells were made from polymethyl methacrylate. Each cell had an internal diameter of 8 cm. The length of the central cell compartment was 10 cm and the length of the electrode compartments was 5 cm. The ion exchange membranes separating the central compartment from the electrode compartments were commercial membranes from Ionics (anion exchange membrane AR204 SZRA B02249C and cation exchange membrane CR67 HUY N12116B). Platinum coated electrodes from Permascand were used. A power supply (Agilent E3612A) was used to maintain a constant current. In each of the electrode compartments, 500 mL of 0.01 M $NaNO_3$ adjusted to pH 2 with HNO_3 was circulated. The soil was kept suspended in distilled water during the experiments by continuous stirring with a plastic-flap attached to a glass-stick and connected to an overhead stirrer (RW11 basic from IKA). The stirring was maintained identical conditions in all experiments.

Conductivity and pH in the soil suspension and the voltage between working electrodes were measured once every 24 h. Due to the electrode processes, pH changed in the electrolytes. The pH in the electrolytes was therefore manually maintained between 1 and 2 by addition of HNO_3 and NaOH. By the end of the electrodialytic experiments, the contents of Cu and As in the different parts of the cell (membranes, soil, solutions, and electrodes) were measured. The suspension from the central compartment was filtered. The sediment was dried and crushed lightly in a mortar by hand before the heavy metal concentrations and pH were measured. The contents of Cu and As in membranes and at the electrodes were measured after extraction in 1 M HNO_3 and 5 M HNO_3, respectively. The energy consumption after treatment can be calculated by equation $E = \int VI dt / W$, where E is the power consumption per gram soil (Wh/g); V, voltage between working electrodes (V); I, current (A); t, duration (h); W, the mass of soil (g).

To investigate the comparison between the original soil and the soil fines and the influence of current intensity and liquid to solid ratio (L/S) on the remediation efficacy of the soil fines, six electrodialytic remediation experiments were performed (Table 1). In Table 1, the L/S 3.5 corresponded to 100 g soil suspended in 350 mL distilled water and L/S 7.0 corresponded to 50 g soil suspended in 350 mL distilled water.

3. Results and discussion

3.1. Soil characteristics

The characteristics of the original soil and the soil fines are listed in Table 2 together with the Danish limiting values for Cu and As for
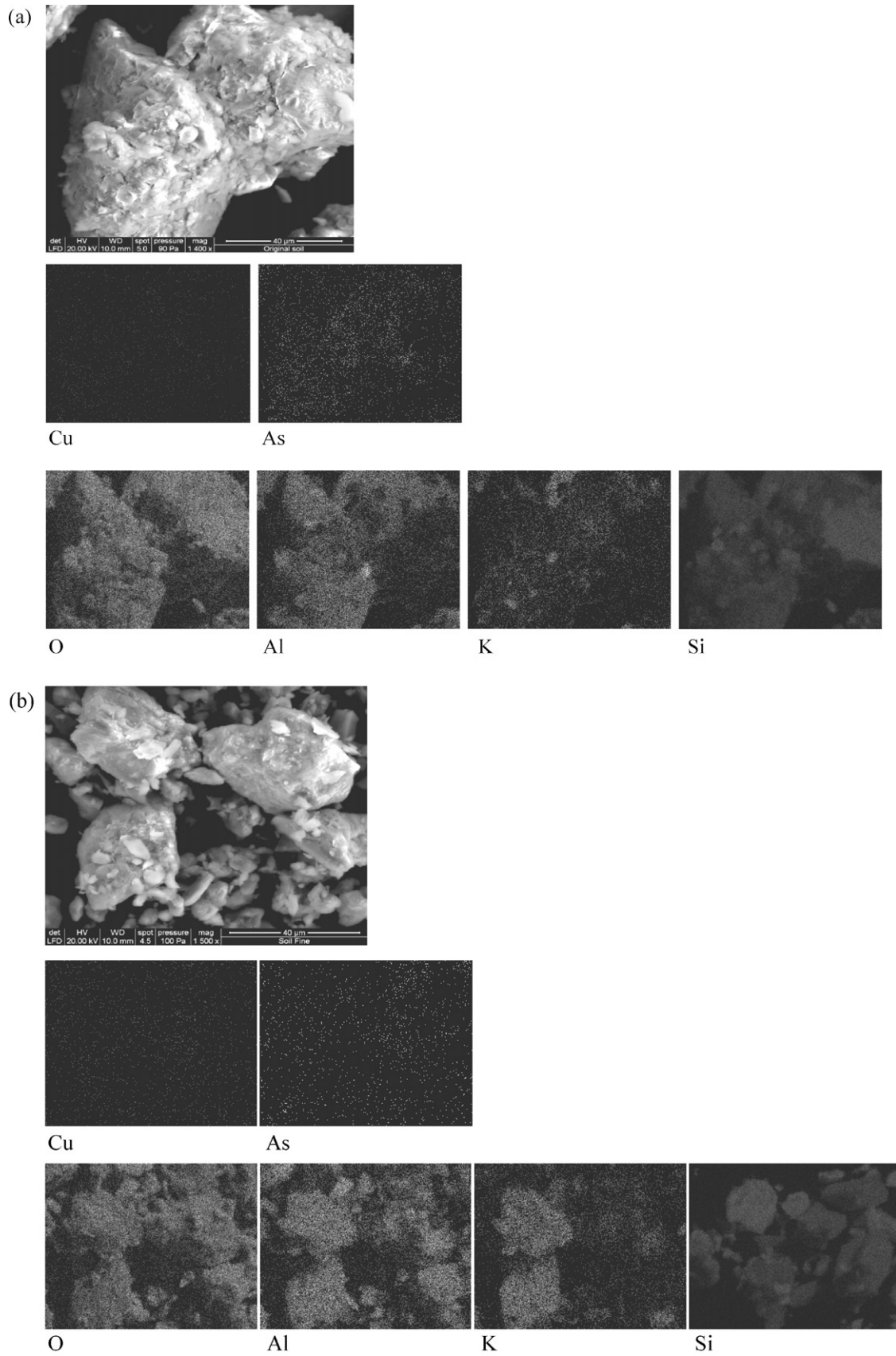
(a)

Cu   As

O   Al   K   Si

(b)

Cu   As

O   Al   K   Si

Fig. 2. SEM picture of original soil (a) and soil fines (b) and result from EDX mapping.

Table 1
Experimental design.

Treatments	Soil type	L/S	Current (mA)	Days
T1	Original soil	3.5	2.5	10
T2	Original soil	3.5	5	10
T3	Soil fines	3.5	5	5
T4	Soil fines	3.5	5	22
T5	Soil fines	3.5	10	10
T6	Soil fines	7.0	5	15

the most sensitive land use. The fine fraction accounted for about 35% of the soil per weight, and more than 90% Cu and 90% As were bound in this fraction, so the soil was suitable for a size fractionation prior to the electrodialytic treatment. Searching the soil samples by SEM-EDX investigation, there was no single particle with high concentrations observed, which might have led to the pollutants being concentrated in the sand fraction rather than in the fine fraction, as was found in [6]. It was found that Cu and As were distributed over the surfaces of the original soil and soil fines (Fig. 2). The low carbonate content in the original soil and the soil fines revealed a low buffering capacity. The pH measured when both the original soil and the soil fines were suspended in KCl was lower than the pH measured in distilled water (0.6–0.8 pH units). This showed that H^+ ions were present in the exchangeable sites at the surface of the original soil and soil fines, since more H^+ ions were released to the liquid in KCl (exchanged with K^+) than in distilled water.

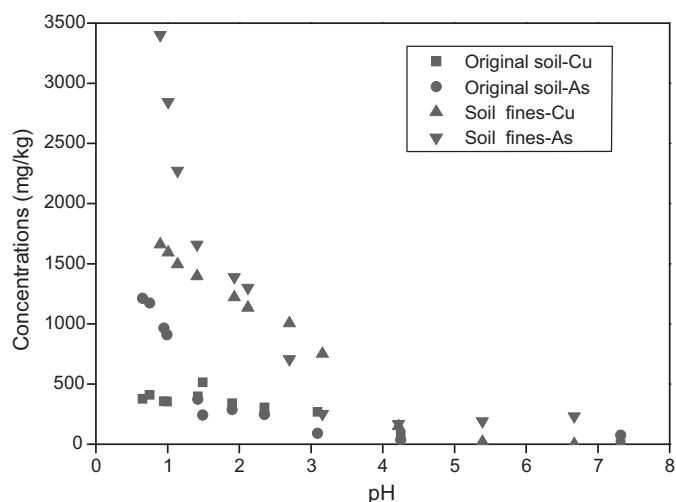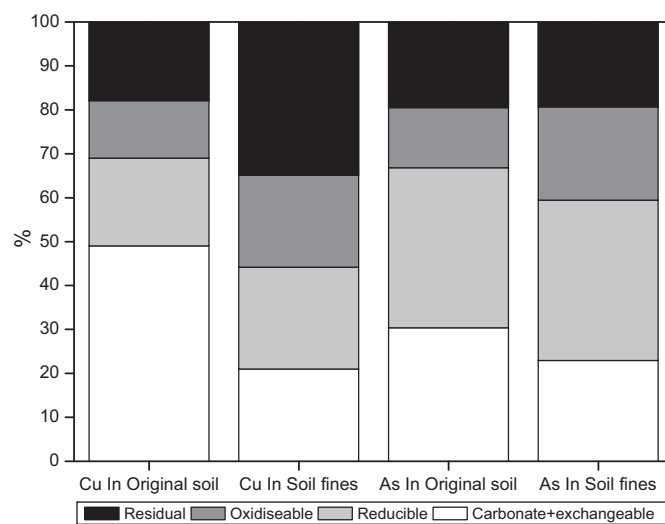### 3.2. Desorption of Cu and As as a function of pH

Fig. 3 shows the concentrations of Cu and As extracted from the original soil and soil fines at different pH values. It was found that the extractions of both Cu and As increased with the decrease in pH. In the original soil, Cu and As extractions started at pH values below about 4.5 and approached 90% and 103% respectively at approximately pH 1. The Cu and As extracted from the soil fines did not reach such high percentages, being 81% for Cu and 74% for As at the same pH level. At neutral pH of the suspension (in distilled water) no measurable amount of Cu was extracted from either the original soil or the soil fines, whereas about 100 mg/kg for the original soil and 200 mg/kg for the soil fines of As were extracted. This does not necessarily indicate that such concentration was present in the soil before the sampling. Mobile As would be expected to have been washed out to deeper soil layers in the 25 years (at least) since the spill occurred. Aeration of the soil sample during sampling and treatment is likely to have influenced the mobility of As.

3.3. Sequential extraction of Cu and As

Fig. 4 shows the result of the sequential extractions of Cu and As from the original soil and the soil fines. For both Cu and As, which showed a similar pattern, the carbonate and exchangeable fraction decreased from the original soil to the soil fines, while in contrast, the residual and oxidisable fractions increased. Cu was adsorbed less strongly in the original soil compared to the soil fines and As showed the same tendency even though not so clearly. The fine fraction was a part of the original soil and thus this result indicates that the small fraction of the two pollutants bound to the coarse fraction in the original soil was bound weakly to the soil particles in

Fig. 3. Desorption dependency on pH of Cu and As in original soil and soil fines.

Fig. 4. Sequential extraction of Cu and As in original soil and soil fines.

comparison to the adsorption in the soil fines. This was consistent with what would be expected when there were no insoluble or poorly soluble precipitates containing the pollutants in the soil (as seen in the SEM-EDX investigation).

3.4. Electrodialytic experiments. Overall results

An overview of the results obtained in the electrodialytic remediation experiments is given in Table 3. The mass balance of an element was defined as the relation between the sum of the mass found in the different parts of the cell at the end of the experiment and the initial mass, calculated on the basis of the measured mean initial concentration. The range of the mass balances was from 94% to 138%, which was an acceptable range for an inhomogeneous industrially polluted soil. The removal efficiency for each element was calculated as the mass of the actual heavy metal in the

Table 2
Characterisation of experimental soil and Danish limiting values for most sensitive land use.

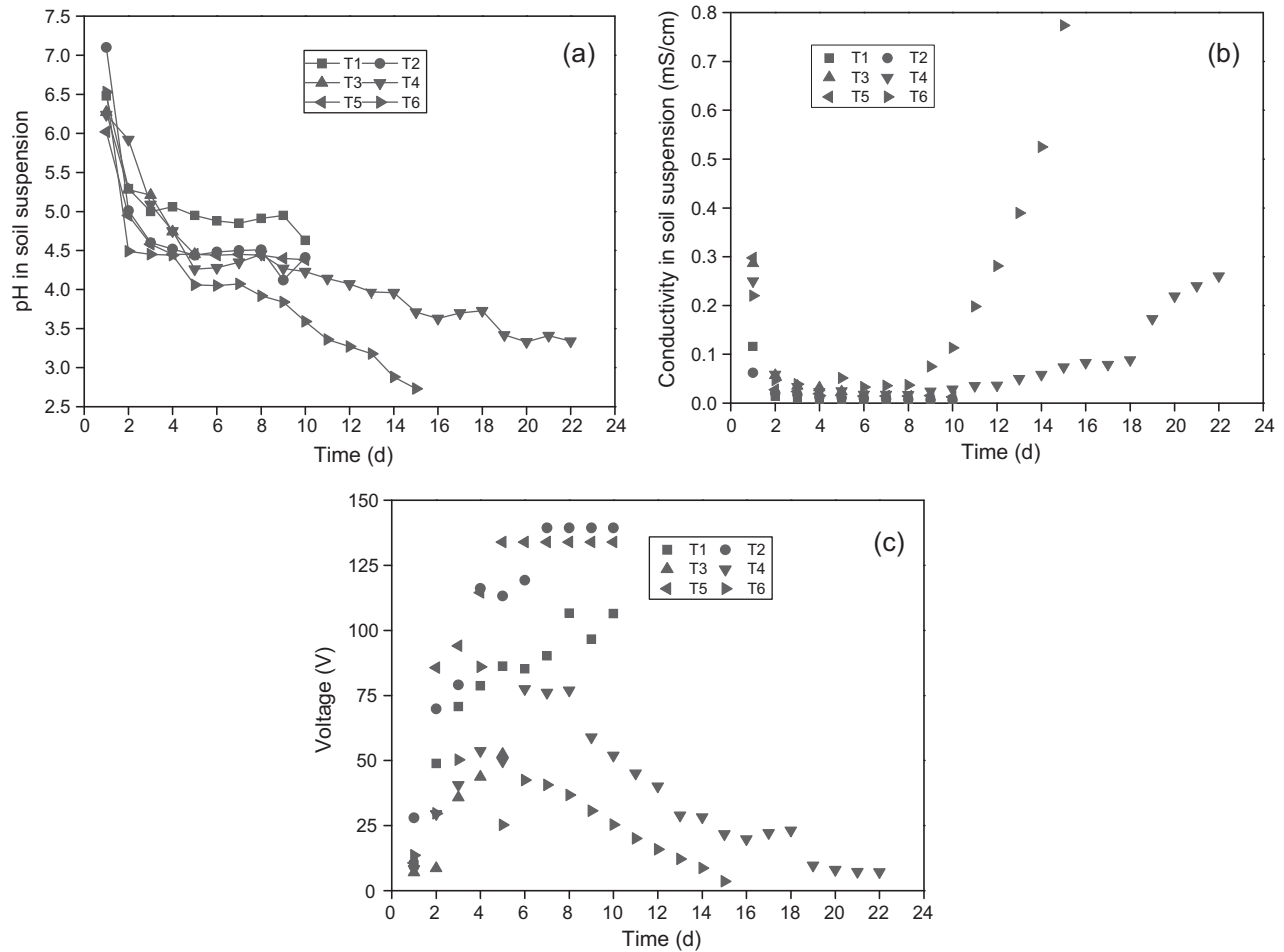|	Cu (mg/kg)	As (mg/kg)	Carbonate content (%)	Organic matter (%)	pH$_{H_2O}$	pH$_{KCl}$
Original soil	573 ± 33	1181 ± 29	0.6 ± 0.2	3.7 ± 1.0	7.4	6.6
Soil fines	2054 ± 62	4598 ± 167	0.2 ± 0.2	3.2 ± 0.8	6.4	5.5
Limiting values	500	20				

Fig. 5. (a) pH and (b) conductivity in the soil suspension and (c) voltage over the cells during the electrodialytic treatment.

electrode components (membranes, solutions in electrode compartments and on electrodes) divided by the total mass found in all parts of the cell at the end of the experiment. Among the treatments, the highest removal percentage was observed in experiment T4 for both Cu (96%) and As (64%). The lowest efficiency for both pollutants was obtained in experiment T6: 13% for Cu and 0.7% for As. The pH measured in KCl was 0.8 to 1.1 pH units lower than the pH measured in distilled water and this was in general a slightly larger difference than was found initially. The difference showed that H^+ ions were present in the exchangeable sites to a higher extent than before eletrodialytic treatment, due to acidification during the treatment. All exchangeable sites were thus not occupied by H^+ ions at the time of sampling.

The pH and conductivity of the soil suspension and the voltage across the cell during the experiments are shown in Fig. 5(a)–(c), respectively. The pH in suspension of all treatments decreased over time (Fig. 5(a)). During the first two days, a considerable drop in the pH of the suspension in the central compartment was seen,

followed by a moderate decrease. In other soils a "lag-period" had been observed before pH decreased in the soil suspension [2]. During the lag-period the H^+ ions overcame the buffering capacity of soil. In the soil of this investigation, the carbonate content was low (both in the original soil and the soil fines) (Table 2), indicating that the soil had a low buffering capacity. The fact that there was a fast drop in the pH of the suspension and that the lag-phase was missing in the present experiments indicates that the lag-phase is dependent on the buffering capacity of the soil. At the end of the experiments, the pH in the soil suspension varied significantly between the different treatments. The rapid acidification of the soil suspension could be due to the water splitting caused by the anion exchange membrane [8,9] and the exchange between H^+ ions from the acidic catholyte and other ions in the suspension over the cation exchange membrane. The pH in the catholyte was maintained between 1 and 2, so this exchange was likely to have a major influence. Moreover there was only 50 g of soil fines in experiment T6, which most probably means the buffering capacity in this soil

Table 3
Overall results of electrodialytic soil remediation.

Treatments	Cu (mg/kg)	As (mg/kg)	pH H$_2$O/KCl	Mass balance (%) Cu/As	Removal efficiency (%) Cu/As	Energy consumption (Wh/g soil)
T1	279 ± 12	600 ± 14	4.6/3.8	112/109	59/56	0.4
T2	367 ± 23	979 ± 30	4.6/3.7	94/138	36/44	1.2
T3	1426 ± 29	3036 ± 42	4.5/3.7	98/97	32/35	0.1
T4	107 ± 6	1715 ± 22	3.3/2.4	118/98	96/64	1.0
T5	1309 ± 35	2534 ± 37	4.5/3.7	105/98	43/47	2.5
T6	2078 ± 92	5529 ± 188	2.7/1.6	113/120	13/0.7	1.0

suspension was even lower than in experiment T4, which had 100 g of soil fines, so that fewer ions were available to exchange with H^+ ions from the catholyte (Table 1).

At the beginning, the conductivity of the soil suspension decreased in all treatments. This could be ascribed to the depletion of free ions in the soil by the applied current (Fig. 5(b)). During the 5–10 day experimental periods, the pH in these treatments did not decrease to below 4.5 (Fig. 5(a)), which suggested that Cu and As had not been desorbed and thus mobilised for electromigration in accordance to the desorption pattern in Fig. 3. In general, the desorption and dissolution processes were expected to be limited at pH 4.5 compared to lower pH values, so the higher pH value in the suspension was one major reason for the lower conductivity, due to a smaller H^+ ion concentration. Further, this might also be the reason for the limited acidification, because a limited amount of cations were available for exchange with H^+ ions from the catholyte. However, in experiments T4 and T6, enough free ions were present in the suspension to overcome this limitation, so the pH decreased to levels where desorption started and was followed by an increase in conductivity.

At the beginning of all the experiments the voltage increased (Fig. 5(c)). In experiments T1 and T3, the voltage increased and did not reach the maximum for the power supply (around 135 V) during the experiments. In experiments T2 and T5 the voltage increased to the maximum of the power supply value and the experiments were stopped on the tenth day. In experiments T4 and T6, the voltage dropped from a maximum on the sixth day and continued to decrease during the remaining time of the experiment. Since the overpotential at the electrodes was low, because the electrolyte was continuously stirred, the voltage increase corresponded to an increase in electrical resistance across the cell. The electrical resistance in the electrolyte compartments was low, due to high ionic strength, so the increased resistance must be across the middle compartment or across the membranes. Initially, when the soil was suspended in distilled water, dissolved ions were removed by the current, resulting in a decrease in electrical conductivity. However, as the acidification started and proceeded, more ions were released and the electrical resistance decreased. This could explain the increase in voltage and also the decrease in voltage in experiments T4 and T6, and the voltage pattern also followed the pH pattern in the soil suspension.

Fig. 6(a) and (b) shows the distribution of Cu and As in the cell at the end of the experiments. Overall the distribution pattern of the two elements differed significantly. In all treatments the major part of the Cu was found either at the cathode or in the soil, whereas for As the major part was distributed either in the anolyte or in the soil. These different patterns indicate the different chemical behaviour of the two elements.

3.5. Comparison of remediation efficacy between original soil and soil fines

The results of experiments T2 and T4, which were performed with the original soil and the soil fines, respectively, with the same L/S and current but different duration, demonstrated a significant difference in remediation efficiency between them. The removal efficiency of Cu and As was 36% and 44% in the original soil against 96% and 64% in the soil fines. In fact, in the experiment with the original soil (T1) it was not possible to maintain the current at 5 mA for more than 10 days, as the resistivity increased until the maximum voltage of the power supply was reached, at which point this experiment was terminated. This means that 5 mA current was too high for this original soil at the actual L/S. However, large amounts of Cu and As were still removed under these conditions. This was possibly due to the large amount of exchangeable species of Cu and As (Fig. 4), which were directly mobile by electromigration [10]. A
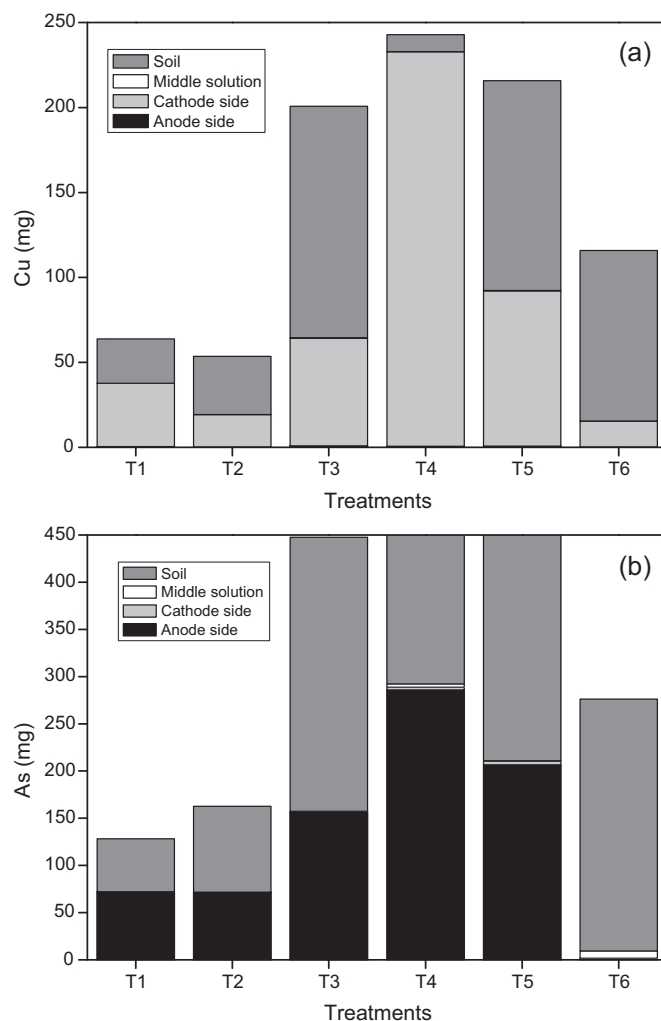

Fig. 6. Distribution of (a) Cu and (b) As in the different parts of the electrodialytic cell at the end of the experiments.

similar experiment performed by Ottosen et al. [6] indicated that constant current could be obtained at the current intensity 2.5 mA in the original soil, although the removal of Cu and As was less than that from the soil fines with 5 mA.

Experiments T1 and T3 were direct comparisons between original soil and soil fines where they had the same mass of charge migration and L/S. In experiment T1 2.5 mA was applied to the original soil to avoid charge overload. In this comparison, the removal efficiency in experiment T1 was much higher than in experiment T3, but the mass of Cu and As removed was reversed: 38 mg Cu and 72 mg As were removed from the original soil and 64 mg Cu and 157 mg As were removed from the soil fines. Moreover, from the viewpoint of energy intensity, the treatment with the soil fines commenced lower energy consumption than the original soil, which was 0.1 Wh/g and 0.4 Wh/g soil, respectively (Table 3).

3.6. Comparison of different conditions for electrodialytic remediation of soil fines

Based on experiences from the work of Jensen et al. [2], two ratios of liquid to solid (3.5 and 7.0), and two values of current intensity (5 and 10 mA, the corresponding current densities were approximately 0.1 and 0.2 mA/cm^2) were investigated. The best conditions in the present investigation were experiment T4 where the L/S ratio was 3.5 and the current intensity was 5 mA. In

experiment T4, an interesting finding was that almost all the Cu and most of the As had been removed from the soil fines even though the pH did not decrease to below 1. In fact, the lowest pH value reached in the soil fines in experiment T4 was 3.5 (Fig. 5(a)). The pH is very important for the desorption and dissolution of heavy metals from soil, because H^+ can destroy the binding forces, induce the change of redox conditions between heavy metals and soil particles and release the heavy metals to solution. These reactions are equilibrium reactions, and they will move to the desorption side with the addition of more H^+ ions. On the other hand, if the products are consumed continuously by other materials or factors (i.e. current), the reactions will also move to the desorption side with the same H^+ concentrations. The constant current enabled continuous desorption due to continuous removal compared to the case where the acidification was performed in a single step (as in the pH desorption experiment). In experiment T4, 96% removal efficiency indicates that the majority of the different adsorbed Cu phases could be removed by the electrodialytic process, including the residual phase, which was expected to be the strongest bound and thus the most difficult to remove (Table 3). This could be mainly attributed to the transformation of heavy metals from the higher fractions of sequential extraction to the first fraction, caused by the combination of acidification and the applied electric field [11]. Not all of the heavy metals in the residual part are bound to the mineral lattice structure, because some of them exist in the form of precipitation and complex compounds, and fortunately they can easily be removed from soil. In general arsenic may be present as As (III) or As (V) in soil as well as in the solution. Which form prevails is dependent on the pH and the redox potential. It may be seen in Fig. 6(b) that a large amount of As was removed towards the anode in this treatment, probably as $H_2AsO_4^-$. Unlike the stationary EKR/EDR, oxygen and carbon dioxide concentrations in suspended EDR can be assumed to be in equilibrium with the atmosphere, which allows for oxidation of As (III) to As (V) during remediation. Further, under the moderately acidic and oxidising conditions created during the process of experiment T4, $H_2AsO_4^-$ should be the prevailing species of arsenic, which would be transferred to the anode side [12].

In experiment T5, the current intensity was increased to 10 mA compared with the 5 mA in experiment T4. However, it was obvious that the free ions in the soil fines were not sufficient to maintain the 10 mA constant current, and induced a higher energy consumption (2.5 Wh/g soil) than T4 (1.0 Wh/g soil). This experiment was an example of the ultimate consequence of forcing too much current through the system: the lack of ions became pronounced (seen from the conductivity of the suspension Fig. 5(b)). As a result, the resistance increased dramatically, and constant current could not be maintained. At the initial stage of the electrodialytic treatment, the current was mainly carried by the soluble and mobile ions from the soil itself. If the conductivity was low but the current was high, the current could not be maintained until desorption and dissolution of ions caused by acidification had taken place, so the process stopped.

The L/S ratio was increased to 7.0 in experiment T6, compared to 3.5 in experiment T4. In experiment T6, the lowest removal efficiency both for Cu and As was observed, while the pH declined to the lowest value and the conductivity attained its highest value at the end of the experiment (Fig. 5(a) and (b)). For Cu, this could be attributed to an expected effect of acidification, which was competition between H^+ ions and Cu^{2+} ions for electromigration. The experiment with the longer acidification time (T4) therefore showed better remediation efficacy than that with very fast acidification (T6), and in the search for optimal remediation conditions, it indicates that the fastest rate of acidification was not optimal. By contrast, the removal efficacy of As was even worse than Cu, which was related to the chemical behaviour of As. This might be because uncharged As species were present and this could very likely be H_3AsO_4 (which was prevailing at high oxidation states and pH values less than about 3 [12]). Another report found that As was only slightly mobile at low pH value, and as a result, it was difficult to remove except in the presence of an enhancement agent [13]. Also due to the fast acidification, the energy consumption in T6 was the same with T4 even under the condition of much less heavy metals removed since the mobility of H^+ ions is high.

4. Conclusions

This paper reports a comparison of Cu and As removal from an original industrially polluted soil and from soil fines from the same soil. The results show that the range of removal efficiency in the original soil and soil fines were from 13% to 96% Cu and 0.7% to 64% As, the highest percentage of removal being from the soil fines in both cases. Among treatments, the highest removal efficiency occurred in soil fines with 5 mA current and with an L/S ratio of 3.5 in a treatment lasting 22 days. In a direct comparison between the original soil and the soil fines with exactly the same charge transfer, 38 mg Cu and 72 mg As were removed from the original soil and 64 mg Cu and 157 mg As were removed from the soil fines. In the suspension of soil fines, a constant current of 10 mA could not be maintained and in the original soil even a current of 5 mA could not be maintained. In treatment with a high L/S of 7.0, acidification took place too rapidly and the pH was very low in the suspension during the whole process, which impeded the transport of Cu and As. The conclusion is that the remediation current and the L/S ratio must both be optimised, as these two parameters are highly dependent on each other.

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