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PAH degradation and redox control in an electrode enhanced sediment cap

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Abstract

Capping is typically used to control contaminant release from the underlying sediments. However, the presence of conventional sediment caps will often eliminate or slow natural degradation that might otherwise occur at the surface sediment. The objective of this study was to explore the potential of a novel reactive capping, an electrode enhanced cap for the remediation of PAH contaminated sediment. The study on electrode enhanced biodegradation of PAH in slurries showed that naphthalene concentration decreased from ~1000 µg/L to ~50 µg/L, and phenanthrene decreased from ~150 µg/L to ~30 µg/L in ElectroBioReactor within 4 days, and the copy numbers of PAH degrading genes increased by almost 2 orders of magnitude. In a cap microcosm, two carbon electrodes were emplaced within a sediment cap with an applied potential of 2 V. The anode was placed at the sediment-cap interface encouraging oxidizing conditions. Oxidation and Reduction Potential (ORP) profiles showed redox potential approximately 60-100 mV higher at the sediment-cap interface with the application of voltage than in controls. Vertical profiles of phenanthrene porewater concentration were obtained by PDMS-coated fiber, and results showed that phenanthrene at the depth of 0-0.5 cm below the anode was degraded to ~70% of the initial concentration within 10 weeks. PAH degrading genes showed an increase of approximately 1 order of magnitude at the same depth. The no power controls showed no degradation of PAH. These findings suggest that electrode enhanced capping can be used to control redox potential, provide microbial electron acceptor, and stimulate PAH degradation.

Keywords

Polycyclic aromatic hydrocarbon (PAH); biodegradation; sediment capping; redox; electrode; electrochemical

Introduction

Contaminated sediment has become a major concern at many sites throughout the United States and the world. A variety of organic and inorganic contaminants such as chlorinated solvents, aromatic hydrocarbons and heavy metals have been found in contaminated sediment and pose a risk to ecology and human health. Because of its high cost and limited

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effectiveness,^{1,2} dredging and ex-situ disposal is sometimes not an efficient method for the treatment of contaminated sediment. In-situ alternatives have significant advantages if they can be demonstrated to be effective. In-situ capping is the placement of clean material (usually sand) on the sediment to isolate the contaminants into the overlying water.^{1, 3} In-situ capping can be a relatively cost-effective and noninvasive approach compared to dredging. However, capping doesn't necessarily provide detoxification, and the risk to ecological and human health may recur if the contaminant can migrate through the caps. Furthermore, conventional capping usually drives the entire sediment anaerobic, hindering natural degradation processes for some contaminants such as polycyclic aromatic hydrocarbons (PAHs).

The use of a reactive capping technology has been proposed to enhance contaminant degradation in capping layers. Unlike conventional capping, reactive caps can facilitate transformations that can detoxify migrating contaminants. The primary difficulty in reactive capping is maintaining conditions conducive to transformations, e.g. redox conditions, sufficient nutrient or electron donor levels.

Electrochemical remediation is among the processes being investigated for their potential application in soil and sediment remediation. Laboratory experiments and limited field work⁴⁻¹² have proven the effectiveness of this novel approach in some soil environments. In most of these reported cases, decontamination of organic contaminant was achieved by direct electrochemical oxidation and/or reduction. Some other successful electro-remediation includes sequestration of heavy metals at the electrode and collection of contaminants by electrokinetic process for further treatment. However, almost all these processes are not cost effective due to the requirement of high voltage input, and they do not utilize the biodegradation potential of indigenous microorganisms in soil/sediment.

A promising alternative approach is to use electrodes as electron acceptor (anode) for microbial communities responsible for PAH degradation. Recent studies show that anaerobic degradation of aromatic hydrocarbons was stimulated by providing an electrode as the electron acceptor.¹³ The application of an electric current in sediment caps using carbon cloth electrodes created engineered redox gradient, suggesting that electrode capping has the potential to stimulate microbial activities for PAH biodegradation.¹⁴

In this study, an electrode enhanced capping was proposed and investigated for the bioremediation of PAH contaminated sediment (Figure 1). In contrast to other electrode based remediation technology, this approach uses low power, and works by encouraging natural biodegradation through the electrochemical process. The application of the electrodes is designed to modify redox conditions in a thin layer in the immediate vicinity of the electrode to encourage biodegradation of contaminants migrating through the cap (as opposed to trying to control redox conditions in the entire contaminated sediment layer).

The potential for this approach was assessed in two types of laboratory scale experiments: 1) slurry reactors under idealized conditions to demonstrate effective enhancement of biodegradation and 2) simulated cap sediments to evaluate the effect of mass transfer limitations under a realistic but controlled laboratory environment. At this early stage in the development of the technology, the focus was on investigating and quantifying the processes involved. Field scale demonstration of this technique was beyond the scope of this study; however, opportunities and challenges to full scale implementation are discussed.

Materials and Methods

Proof of concept testing of electrode enhanced biodegradation was conducted in two types of experiments, slurry experiments and in microcosm experiments at representative sediment densities and conditions. Methods for each are discussed below:

Electrode enhanced biodegradation of PAH in sediment slurry

The electro-biodegradation slurry experiments were conducted using glass electrolysis reactors to minimize adsorption of PAH on the cell wall. Each reactor has a compartment of 500-600 mL volume and several ports for fitting the electrodes into the solution and for sampling.

PAH contaminated sediment was obtained from Anacostia River (Washington DC) and sieved through 2 mm. The sediment was diluted to 1% solid concentration with minimal salt medium (MSM) and then spiked with naphthalene and phenanthrene. Although the Anacostia River sediment contained these compounds, the concentrations were too low to use the sediment without spiking. The presence of these compounds in the sediment, however, ensured that there were likely PAH degraders present that could be enhanced by the technique herein. The minimal salt medium (MSM) contained 4 g of KH_2PO_4 , 4 g of Na_2HPO_4 , 0.5 g of NH_4Cl , 0.2 g of $\text{MgCl}_2 \cdot 6\text{H}_2\text{O}$, and 1 mL of trace element solution SL-4 per 1 L of deionized water. SL-4 was composed of 0.5 g of EDTA, 0.2 g of $\text{FeSO}_4 \cdot 7\text{H}_2\text{O}$ and 100 mL of trace element solution SL-6 per 1 L of deionized water. The composition of SL-6 was 0.3 g of H_3BO_3 , 0.2 g of $\text{CoCl}_2 \cdot 6\text{H}_2\text{O}$, 0.1 g of $\text{ZnSO}_4 \cdot 7\text{H}_2\text{O}$, 0.03 g of $\text{MnCl}_2 \cdot 4\text{H}_2\text{O}$, 0.03 g of $\text{Na}_2\text{MoO}_4 \cdot \text{H}_2\text{O}$, 0.02 g of $\text{NiCl}_2 \cdot 6\text{H}_2\text{O}$ and 0.01 g of $\text{CuCl}_2 \cdot 2\text{H}_2\text{O}$ per 1 L of deionized water. The pH of MSM is 6.8. All preparations were conducted in a nitrogen chamber and the samples maintained under anaerobic conditions.

The following four sets of experiments were conducted:

1. ElectroBioReactor (RE): Power was applied to active sediment slurry that was initial anaerobic;
2. Killed control ElectroBioReactor (KC): Power was applied to killed control sediment slurry (autoclaved) and also initially anaerobic;
3. Aerobic control (AE): No power was applied, but the sediment slurry was maintained in aerobic conditions during the course of the experiment. By maintaining aerobic conditions, the PAH degradation potential of the slurries could be determined;
4. Anaerobic control (AN): No power was applied, but anaerobic conditions were maintained. In this treatment the rate of degradation in an ineffective electrode cap could be determined (that is, one that was ineffective at creating oxidizing conditions).

The applied power was 3.5 V and the current was 15 mA. The slurries were constantly stirred and sampled periodically for PAH analysis. Triplicates of slurry supernatant were collected and extracted using liquid-liquid extraction by hexane for PAH analysis. Hexane was added and mixed with slurry supernatant, and sonicated using a Branson (Danbury, CT) Model 2200 Ultrasonicator for 30 minutes. A 2-mL aliquot of hexane was added in a 5-mL blow down vial. The hexane was then evaporated with a Labconco (Kansas City, MO) Model 79100 RapidVap N_2 Evaporation System to a volume of $\sim 200 \mu\text{L}$ and reconstituted to a volume of 2 mL with acetonitrile. The vial was mixed thoroughly, blown down to the final volume, and analyzed.

The concentration of PAH was measured by HPLC with a 250×4.6 mm column (Phenomenex Luna 5u C18(2) 100A) in a mobile phase composed of 15% water and 85% acetonitrile at 1 mL/min. Detection was made by a fluorescence detector (Waters 2475) with an excitation wave length of 270 nm and an emission wave length of 360 nm.

After the experiment, the slurry was centrifuged at 3500 RPM for 30 minutes and solids were collected and stored at -20 °C. DNA from aliquot of sediment solid was extracted using FastDNA SPIN Kit for Soil (MP Biomedicals). The water content was also measured in order to determine the DNA concentration per weight of dry sediment. The primer set used in quantitative polymerase chain reaction (qPCR) targets aromatic ring-hydroxylating-dioxygenase (RHD_a) for gram negative PAH-degrading bacteria, which covers a large range of functioning genes including *bphA1*, *phnAc*, *nagAc*, *nahAc*, *nahA3*, etc.¹⁵ qPCR was performed in 25 µL reaction volume containing 12.5 µL of Power SYBR green PCR master mix (Applied Biosystems), 2 µL of mixed forward and reverse primer (5 µM), 2.5 µL DNA extract (2.5 µL water for negative control), 2.5 µL of 20X BSA and 5.5 µL water. All DNA samples were analyzed in duplicate or triplicate. qPCR was performed in Applied Biosystems 7900HT sequence Detection System machine. qPCR thermal cycling includes the following steps: step (1) initial dissociation: 95 °C for 5 min; step (2) 40 cycles of the following four steps: 1) Denaturation: 95 °C for 30 s; 2) Primer annealing: 57 °C for 30 s; 3) Elongation: 72 °C for 30 s; 4) Dissociation of the primer's dimers: 80 °C for 10 s; step (3) 72 °C for 7 min; step (4) Melting curve analysis: 0.5 °C temperature increment from 51 °C to 95 °C.

Electro-biodegradation of PAH and redox control in sediment microcosms

Three 400-mL beaker reactors were used to evaluate the ability of carbon electrodes to control the redox potential and encourage PAH degradation in a simulated static sand cap placed over contaminated sediment. One reactor served as a no-applied electricity control (Control) while electricity was applied to the other two (ElecCap1 and ElecCap2) through cloth electrodes in a layer composed of sand and a 10% mixture (mass basis) of siderite to serve as a buffer. In one of the two (ElecCap2), a 0.3-mm layer of siderite was also applied at the sediment and anode interface to aid in buffering.

pH would change at both the anode and cathode due to water electrolysis reactions as follows¹⁶:

Siderite (FeCO₃), once dissolved, could buffer both OH⁻ at the cathode and H⁺ at the anode by the following reactions:

The reactors were filled with sieved (2 mm) sediment (mixture of contaminated Anacostia River sediment and clean University Lake sediment, ratio 1:7). The sediment was spiked with phenanthrene and naphthalene at a level of ~70 µg/g concentration for both compounds. A 5cm×5cm woven carbon cloth (Graphite felt, Wale Apparatus Co, Hellertown, PA) was placed on top of the sediment as the anode. A 3.5-cm layer of sieved (0.425 mm) concrete sand (Riccelli Enterprises, Rush NY) was placed over the anode and a

second, identical, carbon cloth was placed on the sand layer as the cathode. A 0.5 cm sand layer was overlain on the cathode. The sand layers were saturated with electrolyte (2 mM NaCl and 2 mM Na₂SO₄ mixed electrolyte) initially. Water lost due to evaporation was periodically replenished with deionized water to maintain the original salt composition.

The electrodes of both powered reactors were connected by copper wire to 2 V Exttech 382202 DC power supply (Exttech Instruments Corp., Waltham, MA). Power to the powered reactors was continuously applied for ~ 10 weeks except during microelectrode measurement. The anode and cathode of ElecCap2 were switched between day 56 to day 58 and day 66 to day 69 to test if alternating anode and cathode could reverse or reduce pH changes.

The current for ElecCap1 was initially ~200 μ A and gradually dropped to ~100 μ A possibly due to precipitation at the cathode.¹⁷ The current of ElecCap2 was about 200-250 μ A through the entire course of the experiment.

pH was measured by MI-405 standard pH microelectrode (Microelectrodes Inc, Bedford, New Hampshire) and Oxidation Reduction Potential(ORP) was measured by Pt microelectrode against Ag/AgCl reference. Vertical profiles of pH and ORP from the water-sand interface to a depth of 60 mm with 5 mm intervals were acquired and ORP data were converted and reported as versus standard hydrogen electrode (SHE). 2 minutes was allowed to reach equilibrium at each depth.

Phenanthrene profile in the sediment and cap interstitial water was measured by Polydimethylsiloxane (PDMS) coated fibers manufactured by Fiberguide Industries (Stirling, NJ). Naphthalene could not be reliably measured by this technique due to its relatively modest sorption onto PDMS and high volatility. The PDMS-coated fibers had a 210- μ m core with a 10- μ m PDMS coating or outer diameter of 230 μ m. The ability of the PDMS-coated fibers to quantify sediment porewater concentrations has been tested in the laboratory.^{18,19} The linear partitioning relationship of the phenanthrene concentration between the PDMS phase (C_{fiber} , ng/L PDMS) and the porewater (C_w , ng/L water) was assumed with a partition coefficient K_f of 8912.¹⁸

For all PDMS analyses, the method described by Lu¹⁸ was used. The fiber was cleaned prior to deployment by sonication in hexane for a minimum of half an hour, followed by a rinse with acetone and then deionized water. After 24 hours of equilibration of the fibers with the sediment, fibers were rinsed clean (to remove any particles) with deionized water, cut into 1 cm or 0.5 cm pieces, and placed into 2 mL HPLC vials with 1 mL of HPLC-grade acetonitrile. Duplicates of phenanthrene concentration profiles were measured every two weeks during the course of the experiment.

At the end of the experiment, sediment cores were collected and dissected into 0.5 cm or 1 cm long subsamples and stored at -20 °C until further analysis. Naphthalene and phenanthrene concentration of sediment samples was determined using a modified version of EPA Method 3550. Sediment samples were weighed, and mixed with sodium sulfate to disperse the particles and absorb excess water. 60 mL of hexane/acetone (volume ratio 1:1) were added to the jar and sonicated for 30 minutes to extract the PAHs. A 2-mL aliquot of hexane/acetone was added to a 5-mL blow down vial, and then evaporated with a Labconco (Kansas City, MO) Model 79100 RapidVap N₂ Evaporation System to a volume of ~200 μ L and reconstituted to a volume of 2 mL with acetonitrile. The vial was mixed thoroughly, blown down to the final volume, and analyzed. DNA was extracted and qPCR was performed as previously described.

Results and discussion

Biotic and abiotic degradation of PAH in sediment slurry

Figures 2 and 3 depicts naphthalene and phenanthrene concentration in the supernatant of sediment slurry in ElectroBioReactor (RE), killed control (KC), aerobic (AE) and anaerobic (AN) conditions. No significant change was noted for phenanthrene in the KC or AN reactors while naphthalene showed no change in concentration in the AN reactor but a slow first order decrease with time in the KC, suggesting a slow abiotic reaction. The abiotic loss of naphthalene in the electrolytic reactor may be a combined process of direct electrochemical oxidation at the anode, oxidation by hydrogen peroxide, Fenton's reagent and hydroxyl radicals.²⁰

Under aerobic conditions, rapid naphthalene and phenanthrene degradation was noted after a lag phase of about 10 and 30 hours, respectively. Degradation in the ElectroBioReactor (RE) was also rapid but with a slightly longer lag time (about 20 hours and 50 hours for naphthalene and phenanthrene, respectively), and it showed somewhat slower rate than in the aerobic reactor. Evaluation of the number of PAH degrading bacteria in each slurry reactor was consistent with the observed degradation, suggesting biodegradation as the primary reason for the degradation in the ElectroBioReactor. The PAH-RHD_α GN gene levels were normalized by the weight of dry sediment. All the values were reported as an increase from unincubated sediment, which was stored at 4 °C until DNA extraction. Figure 4 shows that PAH degrading gene concentration increased significantly in the electro-biodegradation treatment, in a manner similar to the aerobic reactor, which implied the growth of PAH degrading bacteria. The gene concentration under anaerobic condition was about the same level as that in unincubated control, and this phenomenon was consistent with the fact that PAH was not biodegraded under anaerobic conditions. In the killed control, the PAH degrading gene concentration was not detectable, so it was confirmed that the decrease of PAH in killed control was only due to electrochemical reaction and no biological activities was involved.

Redox control and pH change in sediment microcosms

A more realistic sediment density and the potential for mass transfer limitations was evaluated in the sediment microcosms. Figure 5 shows the observed changes in redox conditions and pH in the microcosms after application of power. The average ORP in the sediment layer near the unpowered anode (d = 4-6 cm) in control reactor was ~90 mv, while those in the powered microcosms, ElecCap1 and ElecCap2, were ~150 mv and ~190 mv, respectively. The increase of ORP in ElecCap1 was less than that of ElecCap2 because of the smaller current in ElecCap1 and consequently smaller proton and oxygen production rate at the anode. Although redox potential increased in the vicinity of the anode, the ORP at the cathode didn't decrease significantly. Redox potentials around the cathode were maintained above 300 mv for all the three reactors. The cathode placement near the interface with the overlying water was sufficient maintain oxidizing conditions near the cathode.

The pH profile in the sand cap demonstrated a similar change in ElecCap reactors (Figure 5). On day 0, the pH between the anode and cathode in all reactors varied from around 7 in the deeper sand/sediment to 9 at the surface. After 2 V potential was applied, the pH gradient through the cap was established: pH in the vicinity of the anode dropped while pH near the cathode increased slightly. pH in the sediment layer of ElecCap1 and ElecCap2 dropped to 6 and 5.7, respectively. The additional buffer in ElecCap2 likely aided in minimizing the pH change in ElecCap2, despite a higher current flow. The pH in the control reactor remained relatively steady with time and depth.

The relatively poor level of pH control using the siderite buffer led to investigation of an alternative means of pH control, switching the anode and cathode of ElecCap2 for a short period (2-3 days) to test the ability to recover pH. Although a pH of 5.7 should not lead to total inhibition of bacterial activity,²¹ neutralization of soil and sediment is generally considered to be favorable for the degradation of mineral oil components by bacteria.²² pH at day 69 was measured after the anode and cathode were switched (anode at $d = 0.5$ cm and cathode at $d = 4$ cm) for 3 days. pH profile at day 69 showed that pH recovery was achieved by this method (Figure 5f). pH around the anode was neutralized to ~6.6 from ~5.7, and pH around the cathode dropped to ~7.2 from ~9. In general, pH was maintained between 6.3 to 7.2 along the vertical profile.

PAH biodegradation in electrode enhanced caps

PAH degradation experiments in electrode enhanced cap simulators were conducted for about 10 weeks. From week 0 to week 6, the PDMS fibers were cut into 1 cm pieces, so the spatial resolution of phenanthrene profile was 1 cm. From week 8 to week 10, spatial resolution was changed to 0.5 cm in the vicinity of the anode. Figure 6 and Figure 7 show the porewater concentration in the cm above and below the anode in the ElecCap cells normalized by that in the control cells as a function of time. The concentration reductions were modest and limited to this small area but some degree of concentration reduction is achieved.

Phenanthrene and naphthalene sediment concentrations in the sediment were quantified at the end of the experiment. Table 1 shows that phenanthrene levels at a depth of 0-0.5 cm below the anode were 77% and 72% of that in the deeper sediment for ElecCap1 and ElecCap2, respectively. This was consistent with the porewater measurement by PDMS fiber. The naphthalene concentrations at a depth of 0-0.5 cm below the anode to that in the deeper sediment were 69% and 46% for ElecCap1 and ElecCap2, respectively. PAH degrading genes were quantified by qPCR for sediment at the depth of 0-0.5 cm below the anode (Figure 8). As in the slurry experiment, an increase of PAH degrading genes was observed in two ElecCap reactors compared to the control reactor. However, the normalized gene copy numbers in ElecCap1 and ElecCap 2 reactors were only 7 and 3 times greater than that in the control reactor. Compared with the gene increase observed in the slurry experiment, i.e., nearly 2 orders of magnitude increase, the gene increases in the ElecCap reactors were much less. In the slurry experiment, the concentrations of phenanthrene and naphthalene decreased rapidly and thus PAH-degrading populations increased significantly. However, in the sediment under the anode, biodegradation occurred slowly. After about 10 weeks, only approximate 30% of phenanthrene was degraded. It is understandable that the growth of PAH degrading bacteria in the sediment was much slower than that in well mixed slurry systems and the control of pH was not as much as had been hoped with the use of siderite, potentially contributing to the lack of more significant difference between control and powered systems.

Conclusions

In this study, the performance of electrode enhanced capping was examined in both an idealized slurry and in a capping microcosm. The results show that electrode enhanced capping can be used to control redox potential in a sediment cap, provide microbial electron acceptor, and stimulate PAH degradation. These findings demonstrated that this novel technology may have applicability for the remediation of contaminated sediment. Although it requires additional placement and maintenance compared to conventional capping, the total cost is still comparable to other active capping techniques. The additional capital costs (compared to capping) mainly include the material cost of electrode materials, i.e. graphite cloth and energy. Energy costs may be minimized through use of wind or solar since energy

demand is minimal ($\sim 0.16 \text{ W/m}^2$, i.e., 1.4 kWh/m^2 per year). Typical costs of large-scale cap applications are approximately $\$17\text{--}\$21/\text{m}^2$ plus the cost of materials.²³ In total, the overall placement cost of electrode enhanced capping is about $\$30/\text{m}^2$, which is similar to the use of a reactive core mat.²⁴ Furthermore, it can be more cost-effective to combine the electrode enhanced capping with a “funnel and gate” system.²⁵ In such a configuration, the contaminants can be funneled by hydraulic control to a relatively small area, or “gate”, and the electrodes only need to be installed at this small gate area instead of for the entire sediment. Severe pH changes at the electrodes could be a potential problem affecting successful application for this novel electrode technology. The addition of a buffer, e.g. as a slowly dissolving solid in the cap layer could help maintain a relative constant pH. Further studies are needed to investigate how to control pH by a solid buffer under in situ flow conditions.

The results obtained in this study show a promising technique to remediate contaminated sediment. For successful application to a specific site, feasibility studies to characterize the site hydraulics and geochemical parameters need to be performed, and operating parameters will likely need to be optimized before scaling up from bench scale to field scale.

Supplementary Material

Refer to Web version on PubMed Central for supplementary material.

Acknowledgments

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Figure 1. Conceptual model for an electrode enhanced cap for PAH remediation. Oxygen is produced at the anode to stimulate PAH biodegradation.

Figure 2. Degradation of naphthalene over time in ElectroBioReactor (RE), killed control (KC), aerobic (AE) and anaerobic (AN) conditions. Points represent the mean of triplicate supernatant samples and error bars indicate standard deviation of triplicate data.

Figure 3. Degradation of phenanthrene over time in ElectroBioReactor (RE), killed control (KC), aerobic (AE) and anaerobic (AN) conditions. Points represent the mean of triplicate supernatant samples and error bars indicate standard deviation of triplicate data.

Figure 4.

PAH degrading gene abundance by qPCR quantification. The PAH-RHD_α GN gene levels were normalized by the weight of dry sediment. All the values were reported as an increase from unincubated sediment control. The results are the means of triplicate, and error bars represent standard deviation. N.D. = Not Detectable.

Figure 5.

Vertical profiles of ORP and pH in control, ElecCap1 and ElecCap2 reactors. ORP values were versus standard hydrogen electrode (SHE). Depth zero was the water-sand interface. Cathode was at $d = 0.5$ cm and anode was at $d = 4$ cm. Sand was between $d = 0$ cm and $d = 4$ cm, and sediment was below $d = 4$ cm.

Figure 6. Phenanthrene porewater concentration in the cm below the anode in the ElecCap cells normalized by that in the control cells as a function of time. The results are the means of duplicate samples or four samples, and error bars represent standard deviation.

Figure 7. Phenanthrene porewater concentration in the cm above the anode in the ElecCap cells normalized by that in the control cells as a function of time. The results are the means of duplicate samples or four samples (error bar data is not shown here for simplicity).

Figure 8.

PAH degrading gene abundance by qPCR quantification at a depth of 0-0.5 cm below the anode in control, ElecCap1 and ElecCap2 reactors. The PAH-RHD_α GN gene levels were normalized by the weight of dry sediment. All the values were reported as an increase from control reactor. The results are the means of triplicate, and error bars represent standard deviation.

Table 1

Phenanthrene and naphthalene concentration in solid phase

Phenanthrene ($\mu\text{g/g}$ dry sediment)	Control	ElecCap1	ElecCap2
C ₀ : 0.5-3cm below anode	69.87 $\pm$ 4.98	78.47 $\pm$ 8.29	66.53 $\pm$ 4.80
C ₁ : 0-0.5cm below anode	61.23 $\pm$ 3.34	60.57 $\pm$ 2.37	47.68 $\pm$ 4.15
C ₁ /C ₀	88%	77%	72%

Naphthalene ($\mu\text{g/g}$ dry sediment)	Control	ElecCap1	ElecCap2
C ₀ : 0.5-3cm below anode	69.65 $\pm$ 13.14	77.39 $\pm$ 8.13	70.09 $\pm$ 18.35
C ₁ : 0-0.5cm below anode	45.75 $\pm$ 8.20	53.56 $\pm$ 3.43	32.58 $\pm$ 0.71
C ₁ /C ₀	66%	69%	46%

Results are average values from triplicate samples, and values after $\pm$ denote standard deviation.