

The Lasagna Technology for In Situ Soil Remediation. 2. Large Field Test

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A full-scale field test of the Lasagna process was conducted at a Department of Energy trichloroethylene-contaminated clay site in Paducah, Kentucky. The test covered an area 21 ft × 30 ft (6.4 m × 9.2 m) and reached 45 ft (13.7 m) deep. A modified sheet piling method was utilized for installing electrodes and treatment zones in thin layers through stiff clay soil without generating solid waste. Iron filings were used in the treatment zones for in situ TCE degradation. Complications encountered during the test included contamination at significantly higher levels than anticipated and complex hydrogeology in the subsurface. Treatment effectiveness seems to vary with location, but in the contaminated areas bracketed by treatment zones, TCE removal ranged from 95% to over 99%. There are strong indications that some of the TCE was transported and degraded in the DNAPL form, which has significant implications. On the basis of the field results, treatment cost for a typical one-acre site with contamination from 15 to 45 ft deep ranges from about \$45 to \$80/yd³. Implemented in its full configuration for the first time at a real site, the Lasagna process has demonstrated its robustness and cost-effectiveness in cleaning up TCE-contaminated soil in place.

Introduction

A novel, in situ remediation technology is being developed for cleaning up contamination in heterogeneous or low-permeability soils. Called Lasagna because of its layered configuration, the technology combines electrokinetic transport of contaminants with in situ treatment methods (1–5). A consortium of industry (Monsanto, DuPont, and General Electric) with collaboration from the U.S. DOE and U.S. EPA have been developing the technology for field implementa-

tion. A low-permeability site at the DOE gaseous diffusion plant in Paducah, KY, was selected for field testing the Lasagna technology. The main organic contaminant at this site is trichloroethylene (TCE), with soil concentrations ranging from below 1 mg/kg (ppm) to approximately 1500 ppm. On a soil weight basis, 220 ppm TCE for this type of soil would be equivalent to pore water saturated with TCE (1100 ppm). Thus, dense nonaqueous phase liquid (DNAPL) is likely present in the area.

An initial small field test called Phase I had been conducted earlier at the Paducah site as a proof of concept of the Lasagna technology under actual field conditions. Reported in Part I of this two-part series (6), the test was completed in May of 1995 after four months of operation. The treatment covered an area 3 m × 5 m and reached 5 m deep. Activated carbon was used as the treatment material to trap the TCE transported from the contaminated soil. Very high removal of TCE from the soil (~99%) was accomplished, even in regions of possible DNAPL. The scale-up from laboratory to this field operation was very successful with respect to key electrical parameters and electroosmotic flow.

A second and larger field test called Phase II was initiated to demonstrate the effectiveness of the Lasagna technology for in situ cleanup of TCE. Also conducted at the Paducah site, this test represented a major step forward from Phase I in that TCE was to be degraded in place and that the treatment was extended to a much lower depth. The dimensions chosen for this field test were 21 ft × 30 ft (6.4 m × 9.2 m) and 45 ft (13.7 m) deep, the depth to which TCE had infiltrated after more than two decades since the original spill. This modular design was intended to be duplicated over larger areas for the actual remediation.

On the basis of extensive data in the literature (7–14) and from our own studies, zero-valent iron in the form of iron filings was chosen as the reagent to degrade TCE. Issues related to TCE degradation in the Lasagna configuration at this site included effectiveness of iron filings in an electroosmotic environment, degradation products of TCE, effectiveness of iron for degrading DNAPL TCE, and role of electrodes in TCE degradation. Other important operational issues were installation of electrodes and treatment zones to the required depth through stiff clay soil without generating solid waste, long-term operation of the process, and extent of soil cleanup as a function of treatment time. Treatment cost, which has been targeted at \$50–100/yd³, was of particular significance in this field test to demonstrate the economic viability of the technology.

A site specific action level of 5.6 mg TCE per kg of soil on the average for the treated area was set by the Commonwealth of Kentucky as the criterion for soil cleanup at this site. This level was based on an assessment model in which the groundwater originating from the site is required to contain less than 5 ppb (the maximum concentration limit for drinking water) by the time it reaches the plant's fence line (15). This action level can be used for this site as long as groundwater monitoring also confirms the modeling results. The test was originally planned to operate for six months. However, with extensive contamination uncovered during the operation, it was extended to one year to allow the process sufficient time to demonstrate its effectiveness in meeting the target cleanup level. This paper summarizes the results of the field test.

Experimental Method

Analytical Procedures. Soil sampling was carried out by Oak Ridge National Laboratory (ORNL) under the direction of

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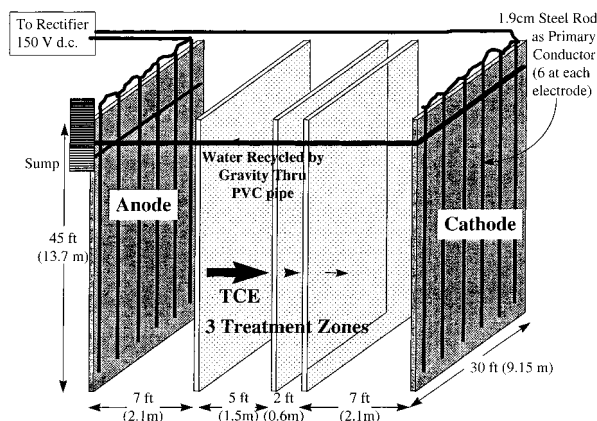


FIGURE 1. Schematic diagram of the field installation.

Martin Marietta Energy Systems (MMES). TCE levels in soil were determined by extracting the soil samples with hexane then analyzing on site using a gas chromatograph equipped with an electron capture detector (GC-EC). Carbon sampling was performed by ORNL and Monsanto. TCE levels adsorbed on carbon were determined by extracting the carbon samples with hexane then analyzing off site (Monsanto analytical laboratory in St. Louis, MO) using GC-EC.

Installation and Operation. The field test covered a soil section 21 ft \times 30 ft (6.4 m \times 9.2 m) and approximately 45 ft (13.7 m) deep. The two electrodes were 21 ft apart, with three treatment zones inserted in between. A passive, underground fluid circulation system was designed to recycle water, which was electroosmotically transported to the cathode, back to the anode. A schematic diagram of the field setup is depicted in Figure 1.

Emplacement Method. The installation was performed by Nilex Corporation of Englewood, CO, under the direction of DuPont and Monsanto. A specially designed mandrel/tremie tube system was used for introducing electrode and treatment zone materials. The hollow mandrel, 51 cm wide \times 5 cm thick and about 17 m long, was made up of 4 tremie tubes, each 12.7 cm wide \times 5 cm, that were welded together. The length of the mandrel allowed it to be driven 14 m into the ground with some extra length remaining above ground. A crane and a vibratory hammer were used to move and drive the mandrel, which was mounted on an 18 m mast tower. Materials needed for each insertion were mixed in a cement mixer and delivered to the hopper that was fastened to the top end of the mandrel. The drives were overlapped by 12.7 cm (1 whole tremie) to ensure continuity of the electrode and treatment zones.

Electrodes and Treatment Zones. Each electrode comprised a 1.5 in. (3.8 cm) thick curtain of 50/50 by volume iron filings and Loresco coke emplaced to a depth of 45 ft (13.7 m). Loresco coke is sold commercially as a backfill material for cathodic protection of buried metal objects such as pipes and tanks. Iron filings were added to promote iron corrosion over water electrolysis as the predominant reaction at the anode, thus minimizing acid (H^+) formation that would reduce the effectiveness of electroosmosis. Six steel rods (1.9 cm diameter, hot rolled) were inserted approximately 1.5 m apart into the electrode materials to a depth of 12.2 m for electrical connections. They were connected in parallel to the main power cable laid along the ground using cathodic protection welding techniques. Electrical conductivity of the installed electrodes measured between the outermost rods gave a value of 4.5 S/m, which was 200 times higher than the soil conductivity.

Three treatment zones, each 1.5 in. (3.8 cm) thick, were installed between the two electrodes. The first treatment zone was 7 ft (2.1 m) from the anode, the second one 5 ft (1.5 m)

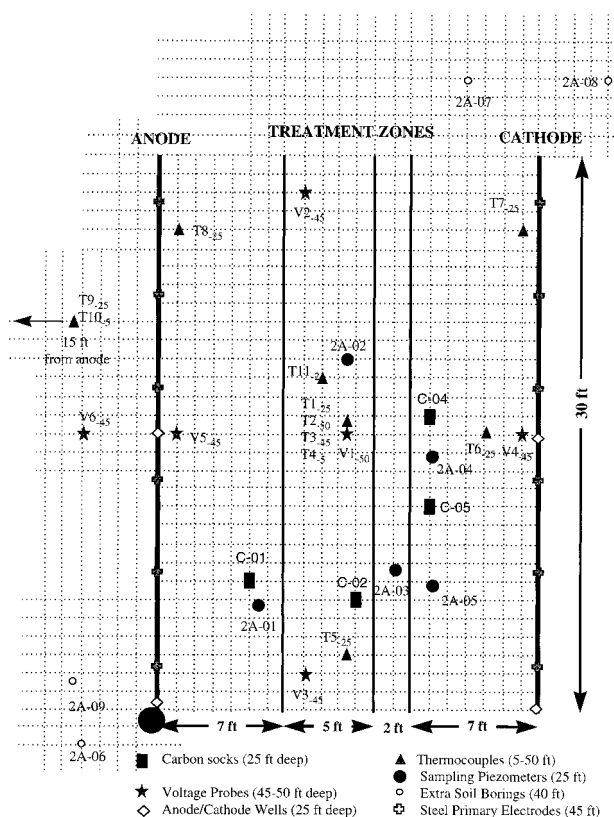


FIGURE 2. Locations of monitoring points.

from the first one, and the last one 2 ft (0.6 m) from the second one and 7 ft (2.1 m) from the cathode (Figure 1). These variable spacings were intended to help determine the optimum spacing as well as to provide information on soil conditions at various stages of treatment. Smaller treatment zone spacings would result in shorter treatment times but higher material and installation costs.

Treatment zone materials consisted of a slurry of Peerless iron filings suspended in wet kaolin clay. Clay was used to help minimize TCE loss through volatilization and provide a stabilized medium for keeping the iron well dispersed throughout the 14-m depth. The iron filings used were of 8–50 mesh size with a BET surface area of 1.6 m²/g and contained about 3% carbon. The actual amount of iron filings in the clay slurry was 26 vol % (equivalent to 1.61 g Fe/cm³). All materials chosen for electrodes and treatment zones are innocuous to the environment and are designed to be left in place after the cleanup is completed. This is an important consideration in the implementation of the Lasagna technology to minimize both treatment costs and disturbance to the environment.

Monitoring Probes. The voltage, temperature, water, and carbon monitoring systems were installed with the Nilex equipment using a 10 cm \times 10 cm square tube tremie that was about 15 m long. A total of 12 thermocouples and six multi-depth voltage probes were installed in the unit. The thermocouples were installed at various depths and locations to monitor the temperature rise in the soil due to resistive heating. Each voltage probe measured the voltage potential at 1.5 m intervals vertically. The carbon socks were 7.6 m long and contained granular activated carbon. A plastic tie-wrap was fastened around each sock at 30 cm intervals to enable discrete measurement of TCE concentrations as a function of depth. Several segments of the carbon socks were later filled with bentonite clay to prohibit any vertical TCE migration in the wells. Figure 2 shows the locations of the installed monitoring equipment and piezometers.

Water Management System. Overflow “wells” installed in two of the cathode drives allowed the water collected at the cathode to be transferred by horizontal pipes to a sump for recycle back to the anode. The anode also had “wells” installed in two drives, which were connected to the sump. To help keep the water handling systems from freezing, the horizontal piping was installed below grade and the sump. The cathode side of the system was placed 1.4 m below grade and the anode piping was installed 1.5 m below grade. The anode was kept at a lower level to ensure the cathode fluid would drain to the sump and anode area. The drop in water levels enabled measurement of the liquid flow rate as water drained out of the cathode piping into the sump; this setup also prevented electrical shorting between anode and cathode through the water conduit.

Electrical System. Power to the test site was supplied with a diesel-run Olympian power generator CT-110 rated at 100 kW. A Rapid Power Technology model SCRA 410 C425 rectifier provided the DC power to the field unit. About 50 ft of heavy copper wire cable (0000 size) was used to connect the power rectifier to each electrode.

Soil Sample Locations. Five soil sampling locations were chosen on the basis of the initial experimental design. Shown in Figure 2, 2A-01 was placed about 1.5 ft (46 cm) upgradient of the first treatment zone. Location 2A-02 was between the first and the second treatment zone (5 ft spacing). Location 2A-03 was between the second and the third treatment zone (2 ft spacing), and 2A-04 and 2A-05 were about 1 ft (30 cm) behind the third treatment zone. Samples were collected by taking 1.2 m long soil cores from the sampling hole. Each core was then subcored to obtain soil samples at every 30-cm interval. Subsequent samples (intermediate and post-treatment) were taken by shifting about 30 cm to either side of the original boreholes in a parallel direction to the electrodes and treatment zones.

Electroosmotic Permeability Calculation. The electroosmotic permeability can be calculated using the following equation:

$$Q = k_e \Delta E A$$

where Q = electroosmosis volumetric flow rate (cm^3/s); k_e = coefficient of electroosmotic permeability ($\text{cm}^2/\text{V s}^{-1}$); ΔE = voltage gradient applied across the soil mass (V/cm); A = cross-sectional area perpendicular to the direction of flow (cm^2).

The electroosmotic flow rate was measured by timing the amount of water flowing from the cathode pipe into the sump, which was caused by electroosmosis since the flow quickly slowed then stopped when the power was turned off. In case of significant rainfall, flow rate measurements were not carried out until several days after the rain had stopped, to minimize contributions from anomalous sources.

Results and Discussion

Site and Soil Characteristics. The area at the Paducah gaseous diffusion plant cylinder drop-test site is mostly clay with gravel and clay overburden and has been used as a truck road. The soil properties have been reported earlier (6). Cone-penetrometry tests were conducted at three locations in the vicinity of the test area to approximately 50 ft deep. It appears that the water table is not present until about 30 to 40 ft. However, there are saturated zones above this at various depths. On the basis of lithology logs, the water is probably in the coarser zones that reside on top of the tight clay zones.

The contaminant at the site is primarily TCE. In preparation for this field test, nine soil borings were made to confirm the northern edge of the plume and to determine the baseline of TCE concentration in the test area. TCE concentration

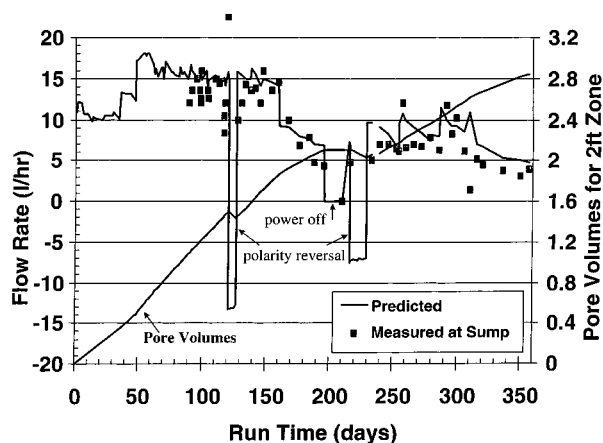


FIGURE 3. Electroosmotic flow rates and accumulated pore volumes as a function of operating time.

ranged from below 1 mg/kg (ppm) to approximately 1500 ppm and appeared to taper off to below detection limits at about 9–11 m deep. On the basis of these soil data, a decision was made to extend the treatment to a depth of 14 m to ensure complete cleanup of the site. With a soil density of $2 \text{ g}/\text{cm}^3$ and 40% porosity ($\sim 20 \text{ wt } \%$ moisture at saturation), a TCE soil concentration of 220 ppm would result in pore water saturated with TCE (1100 mg/kg). Thus, a level of 1500 ppm is a definite indication of the presence of DNAPL.

Electrokinetic Effects. The field installation was carried out between June and August of 1996. The test was started (power on) on August 18, 1996, and lasted almost 1 year. There was a 3-week downtime in March 1997 for intermediate soil sampling, and there were two polarity reversal periods, the first one for 5 days and the second 10 days.

The total voltage was set initially at 150 V ($\sim 0.23 \text{ V}/\text{cm}$) with a corresponding current of 110 amps, designed to limit the maximum temperature rise in the soil to about 60°C (see discussion on thermal effects below). However, when the early water samples showed the extraordinarily high levels of TCE in almost all of the wells, the voltage was raised to 200 V with a corresponding current of 200 amps on October 1, 1996, to accelerate both electroosmosis and soil heating. After 160 days from the start, when the core temperature at the center of the soil mass (hottest spot) was over 80°C , the power was reduced to 120 V and 120 amps to keep the soil from getting too hot. The less-than-proportional increase in the voltage at the higher current was due to the soil becoming more electrically conductive when heated (4).

Voltage distribution in the soil between the two electrodes was reasonably uniform at the beginning of the test. After about five months of operation, however, the voltage profile at the anode started to deteriorate. For some reason, the water distribution wells at the anode were filled up with clay, preventing the recycle water from rewetting the anode and the nearby soil regions. Once the electrode started to dry out, the resistance would go up with temperature, causing more drying. During the March soil sampling, three new anode wells were installed as an attempt to remedy this problem. The voltage distribution became better after the addition of the new wells, but was not as good as at the beginning. In the last few months of the test, because of the poor electrical contact experienced at the anode region, the voltage was raised to about 150–180 V to maintain the same current.

Water flow rates measured throughout the test period correlate well with the predicted flow rates based on the electroosmotic permeability determined in laboratory experiments and from the previous field test ($k_e = 1.2 \times 10^{-5} \text{ cm}^2/\text{V s}^{-1}$). Figure 3 shows the measured and predicted flow

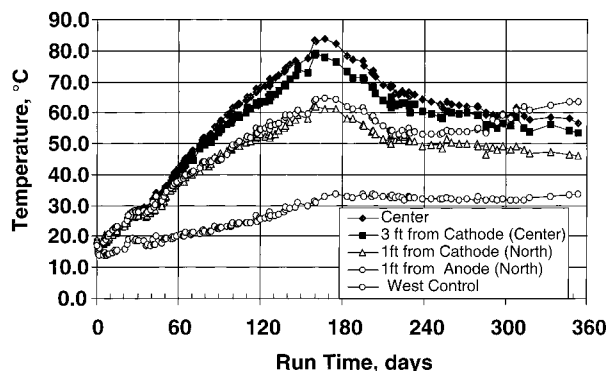


FIGURE 4. Temperature trends at 25 ft (7.6 m) depth.

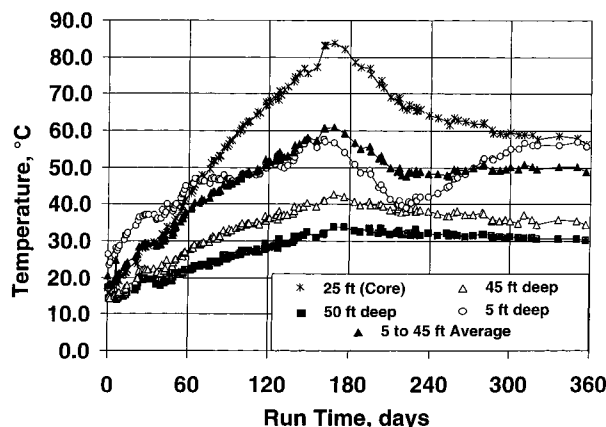


FIGURE 5. Temperature profiles at center of test site.

rates and the cumulative pore volumes of water passed through the system by electroosmosis. Note the three-week shutdown at about day 200 for the intermediate soil sampling and the two polarity reversal periods that were implemented in an effort to rewet the anode region. The total distance the water front traveled during the experiment was approximately 1.7 m, equivalent to 2.8 pore volumes of the soil between the 2 ft (61 cm) treatment zone spacing. The cathode water (and the sump) quickly became alkaline (pH ~ 12) as expected, and the conductivity rose to over 15 000 $\mu\text{S}/\text{cm}$. Representative samples of the anode water could not be obtained because the sampling port was too close to the recirculating cathode fluid.

Thermal Effects. The electrokinetic and thermal model developed in laboratory studies (4) and refined in the Phase I field test (6) was used to predict the thermal behavior and suggest operating conditions for Phase II. Details regarding the utilization of this model in the design of Phase II operations can be found in a recent DOE report (20). The soil temperature played a key role in this field test in several respects. First, elevated temperatures caused an increase in the rate of TCE degradation with iron filings. Second, the diffusion and dissolution rates of TCE in water would increase with temperature, which would result in accelerated TCE removal from the soil matrix by electroosmosis.

Figure 4 shows the temperature trends at the 25-ft (7.6-m) depth across the site. There was a reasonable symmetry in the temperature profile with respect to the two electrodes for most of the duration of the test. In the last few months, however, the temperature in the anode region rose more rapidly than its counterpart at the cathode, probably because of soil dehydration resulting in increased resistive heating. Figure 5 shows the temperature trends at the center of the unit at various depths. As can be seen, the temperature at the center 25-ft depth (7.6 m) shows a maximum temperature of 83 °C at day 160, then dropped to a final temperature of

less than 60 °C at the end of the test. The estimated average soil temperature between 5 and 45 ft increased from 18 °C to a peak of 60 °C between months 5 and 6, then settled down to about 50 °C for the rest of the time. It is not completely clear why the temperature near the surface (5 ft or 1.5 m deep) went up anomalously in the last few month of the test. It could be that this period happened to coincide with the summer months, during which time the ground temperature went up significantly.

Overall, the field operation ran fairly well, with electrokinetic results in good agreement with previous data. The problem encountered with the anode operation was due primarily to the inefficient water distribution system, which failed to keep the anode sufficiently moist for optimum electrical distribution. This is a key learning point from the test, and an improved water distribution system was incorporated in the design for the full-scale cleanup.

TCE Removal. The dechlorination of TCE using zero-valent iron has been studied extensively by many research groups (7–14). The effectiveness of this approach has been demonstrated either as permeable reactive walls or packed bed reactors at many field sites (15–17). The dechlorination apparently goes through a stepwise process in which one chlorine atom is removed each time:



where R stands for hydrocarbons. According to this mechanism, TCE dechlorination products would occur in the following order: c-dichloroethylene (c-DCE), vinyl chloride (VC), ethylene, and ethane. However, while ethylene and ethane are readily observed as reaction products, very little of the intermediates DCE and VC could be detected (typically about 2% of the initial TCE level), probably because they remain bound to the iron surface and undergo further dechlorination.

Dechlorination Rates and Design Criteria for Treatment Zones. The design of the treatment zones took into consideration the required residence time for complete destruction of TCE and its daughter products, especially VC, optimum level of iron with respect to material cost and long-term performance, ease of installation, and minimization of TCE loss through volatilization in the treatment zones.

The rates of TCE dechlorination were measured using the Paducah groundwater both in batch (stirred tank) and continuous (packed bed) mode with Peerless iron. With a packing density of 2.41 g/cm³ and packing porosity of 0.614 for the iron filings, a first-order rate constant of 0.31 h⁻¹ was obtained at room temperature, equivalent to a half-life of 2.2 h. TCE daughter products were found to degrade more slowly: 5.2 h half-life for c-DCE and 3.3 h for VC (20).

At the operating voltage gradient of 0.25 V/cm, water would move at an average speed of 0.43 cm/day through the 3.8 cm thick treatment zone (kaolin clay, 60% porosity), which resulted in a residence time of 212 h. This would be much more than needed for complete destruction of TCE and its daughter products. Even at a very low iron concentration of 8 vol % in clay (0.5 g Fe/cm³), practically complete conversions of TCE, c-DCE, and VC would occur in a single pass (Tables 1 and 2). Also listed in Table 2 are the calculated concentrations of these compounds in the water exiting the treatment zone for a hypothetical pore water containing 1000 ppm TCE, 10 ppm c-DCE, and 10 ppm VC. With the actual amount of iron in the treatment zones at 26 vol % (1.61 g Fe/cm³), the treated soil at 40 to 60 °C for most of the operation time (which increases the rates three- to four-fold relative to the room temperature rates), and 2 pore volumes as the design criterion (contaminants passing through the treatment zones twice), the treatment zones were basically oversized by

TABLE 1. Half-Lives of TCE, c-DCE, and VC as a Function of Temperature. Dechlorination with Peerless Iron Filings at 0.5 g Fe/cm³

T °C	half-life (h)		
	TCE	c-DCE	VC
25	10.7	25.3	16.2
40	3.5	8.2	5.2
60	2.4	5.5	3.6

TABLE 2. Dechlorination of TCE Passing Through a Treatment Zone Packed with 8 vol% Peerless Iron (0.5g Fe/cm³)

T °C	% conversion ^a			effluent concn ^b (ppm)		
	TCE	c-DCE	VC	TCE	c-DCE	VC
25	99.99	99.25	99.90	8.02E-02	7.54E-02	9.61E-03
40	100.00	100.00	100.00	1.48E-06	1.04E-04	1.72E-06
60	100.00	100.00	100.00	5.84E-09	3.21E-06	2.02E-08

^a Assuming plug flow with axial dispersion. ^b Inlet: 1000 ppm TCE; 10 ppm each for c-DCE and VC.

an order of magnitude to ensure complete destruction of the organic contaminants.

A major unknown was the fate of TCE in DNAPL form, which was likely present at various spots in the test site. Laboratory experiments were therefore conducted to investigate the dechlorination of DNAPL TCE. It was found that as long as some water was present (as low as 0.6% in pure TCE), the degradation of TCE in the DNAPL form could occur. At a TCE concentration of about 6000 ppm, which is more than five times its solubility limit at room temperature, the rate followed zero-order kinetics and was about 1/20 of the dechlorination rate of dissolved TCE at 1000 ppm. An unexpected outcome was the product distribution. For dissolved TCE, the products observed were primarily ethylene and ethane in equal amounts, with very little acetylene. A simulated DNAPL situation, however, led to acetylene as the major dechlorination product, followed by ethylene, with very little ethane (19). This difference in product distribution provides an excellent marker for determining TCE degradation in the field.

Carbon Sock Data and Evidence of DNAPL Movement. The four carbon socks inserted at various soil regions (C-01, -02, -04, and -05 in Figure 2) in the test site were intended for periodic monitoring of the accumulated movement of TCE in the soil. The first set of carbon socks was recovered after three months of operation and analyzed for TCE and its main daughter products. (Table 3). Note that TCE distribution is consistent with soil sampling data showing high TCE contamination in the 15–25 ft (5–8 m) depth regions. One intriguing aspect of these results is the very high levels of TCE found on the carbon. The simple calculation shown at the bottom of Table 3 illustrates that even if the soil pore water were 100% saturated with TCE (equivalent to 220 ppm in soil), the amounts of TCE trapped on the carbon for all four socks from the 17 to 23 ft (5.2–7 m) depths were many times higher than the total dissolved TCE present in the pore water passing through. Data at the 4- and 5-month times (not shown) revealed even more accelerated accumulation of TCE on the carbon, probably a consequence of higher soil temperatures.

These carbon results show clearly that the amounts of TCE in the soil were much higher than the average 50 ppm indicated by the pretest soil sampling. More importantly, some form of DNAPL transport must have taken place in order to explain the carbon data. This is a significant finding because it means that the total treatment time could be

TABLE 3. Results from First Carbon Sock Samples^a (Assuming 100% TCE Extraction Efficiency in One Extraction with Hexane)

depth (ft.)	TCE on carbon (ppm) ^b			
	C-01	C-02	C-04	C-05
4		83		2
5		52	20	7
6	12	55	18	19
7	17	77	80	36
8	16	22	26	20
9	28	83	26	18
10	23	5265	52	23
11	69	9306	33	27
12	2,064	6562	11	65
13	6,937	5595	11	161
14	9,465	8199	80	369
15	14,714	24392	115	441
16	29,704	41,436	163	816
17	81,420	105,190	100	2,715
18	116,989	136,687	1,747	3,441
19	118,259	154,094	54,230	9,029
20	126,433	169,319	31,377	24,221
21	102,790	150,946	53,571	18,591
22	117,045	80,521	20,264	92,201
23	127,156	99,562	140,783	151,242
total TCE on each carbon sock (g)	123	144	49	44

^a Carbon socks first sampling, Nov 13, 1996. Operation time: three months. ^b Each carbon sock is 3.8 cm in diameter, contains 144 g wet carbon/ft (30.5 cm). The sock sectional area per ft in direction of flow is 116 cm². At an EO flow velocity of 0.8 cm/day, the amount of water flowing into each sock section is equal to 116 cm² × 0.8 cm/d × 0.4 (soil porosity) = 37.1 cm³/day. If water is saturated with TCE (1100 ppm), the amount of TCE absorbed per ft of carbon sock for three months is 1100 mg/L × 37.1 cm³/day × 0.001 L/cm³ × 90 days = 3.67 g TCE. Thus, average TCE loading on carbon for the three-month time is 3.67 g TCE/144 g carbon = 25,500 ppm.

significantly shorter than expected on the basis of TCE transport as dissolved in water. To check this possibility, laboratory experiments were conducted at room temperature with clay soil containing TCE at concentrations above saturation. This simulated nonaqueous phase TCE was found to move toward the cathode at a rate of about 1/6 to 1/10 of the electroosmotic flow (20). A possible mechanism for this transport is that the clay is hydrated with several layers of water molecules which surround the DNAPL and drag it along under the electric field. At the elevated temperatures of the field, additional mechanisms could be at work for enhancing DNAPL transport, such as a higher dissolution rate of TCE droplets in the soil pores and rapid vapor diffusion of the volatilized TCE throughout the vapor space of the soil matrix followed by its reabsorption into the pore water.

Dechlorination Results from Water Samples. Water samples from various wells within the test site and from a control well (MW 160) were taken several times during the test. The results are shown in Table 4. It can be seen immediately that, throughout the test, samples from the control well showed high levels of TCE but nondetectable levels of the main product gases (ethylene, ethane, and acetylene). In contrast, all of the samples from wells in the test zone showed high levels of the three product gases. For example, the control sample on Dec 13, 1996, showed 73 ppm TCE and none of the other gases, whereas well 2A-03 had a comparable TCE level (78 ppm) but with 144 ppb acetylene, 466 ppb ethane, and 223 ppb ethylene. The same is true for the intermediate samples and for the final samples taken in July. It is thus clear that dechlorination of TCE was definitely occurring in the test zone but not in the control area.

Additionally, the pattern of product distribution seemed very consistent with the combination of the two pathways

TABLE 4. TCE and Its Gaseous Daughter Product Profiles from Water Samples

date	sample	TCE and degradation product gases (ppb)				C2 Ratios	
		acetylene	ethane	ethylene	TCE	acetylene:ethylene	ethane:ethylene
	dissolved TCE					< 0.1	1.0
	DNAPL TCE					> 10	0.2
12/3/96	well 2A-01	239	109	140	na ^a	1.7	0.7
	well 2A-03	149	405	177	na	0.8	2.3
	well 2A-04	74	289	198	na	0.4	1.5
	well 2A-05	15	385	113	na	0.1	3.4
	cathode	426	38	168	na	2.5	0.2
12/13/96	MW 160 (control)	nd ^b	nd	nd	73 000		
	well 2A-03	144	466	233	78 000	0.6	2.1
	well 2A-04	61	262	153	390 000	0.4	1.7
	well 2A-05	33	379	165	850 000	0.2	2.3
6/6/97	MW 160 (control)	nd	nd	nd	61 000		
	well 2A-03	59	629	311	45 000	0.2	2
	well 2A-04	49	306	94	81 000	0.5	3.3
	well 2A-05	49	720	297	490 000	0.2	2.4
	cathode	126	43	89	2500	1.4	0.5
7/9/97	MW 160 (control)	nd	nd	nd	61 000		
	well 2A-03	58	816	385	47 000	0.07	2.1
	well 2A-04	37	236	66	76 000	0.16	3.6
	well 2A-05	91	763	290	280,000	0.12	2.6
	cathode	183	123	164	2300	1.5	0.75

^a na: not measured. ^b nd: non-detect.

TABLE 5. Summary of TCE Analysis of Soil Samples

sample location	TCE in soil (ppm)			total TCE removal (%)	treatment zone spacing (ft)	number of pore volumes ^c
	pretest (3/96)	intermediate (3/97)	final (8/97)			
2A-01						
min	0.106	nd	nd			
max	148	1,504	106			
ave	42	90	24	42.9	7 ^a	0.8
2A-02						
min	nd		nd			
max	125		10.6			
ave	18.4		0.89	95.2	5	1.1
2A-03						
min	nd	nd	nd			
max	309	27.4	1.48			
ave	50.8	2.20	0.15	99.7	2	2.8
2A-04						
min	nd	nd	nd			
max	152	65.8	47.9			
ave	33.6	12.9	9.46	71.8	7 ^b	0.8
2A-05						
min	0.188	0.050	nd			
max	123	116	64.4			
ave	31.4	16.4	9.19	70.7	7 ^b	0.8

^a Distance from the anode to its adjacent treatment zone. ^b Distance from the cathode to its adjacent treatment zone. ^c Based on pore volume of the soil between two treatment zones or between a treatment zone and an electrode.

described above for TCE dechlorination. High levels of acetylene, which suggest that dechlorination of DNAPL was dominating, accompanied intermediate levels of ethylene and much lower levels of ethane. On the other hand, low levels of acetylene, which suggest that dechlorination of dissolved TCE was dominating, accompanied much higher levels of ethane and intermediate to comparable levels of ethylene. It should be noted that high levels of the three product gases were also detected in the cathode water samples, indicating TCE dechlorination was occurring there as well. This cathodic degradation of TCE has been observed in our laboratory studies.

Soil Data and TCE Removal Estimates. While the carbon data and dechlorination results from the water samples were interesting, the deciding factor in judging whether the

Lasagna process was successful in cleaning up the site rested on actual soil samples. As mentioned earlier, the Commonwealth of Kentucky has set a clean up target of 5.6 mg of TCE per kg of soil on the basis of modeling assessment (15). The test would be considered successful if the average of the soil samples in the treated area was less than this level. Results of the three soil sampling events—pretest, mid-test, and post-test—are plotted in Figures 6–10 and summarized in Table 5 in terms of TCE removal for the five sample locations. As described in the Experimental Methods section, soil samples were taken every 1 ft (30 cm) at each borehole from 5 to 45 ft (1.5–12.2 m) deep.

The pretest soil sampling, performed in March 1996, shows that TCE concentrations in the soil ranged from nondetectable to over 300 ppm, with the bulk of contamination

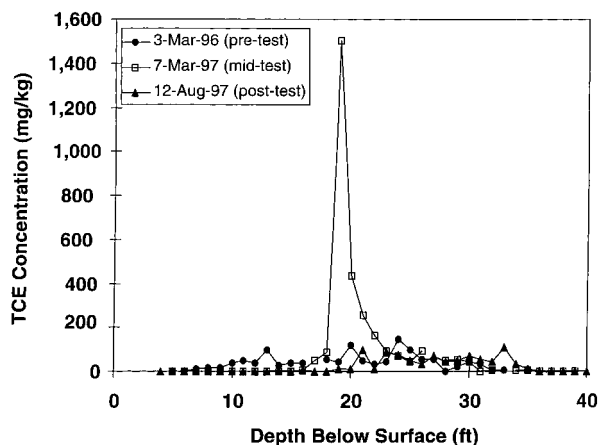


FIGURE 6. Soil TCE concentration profile as a function of depth for location 2A-01.

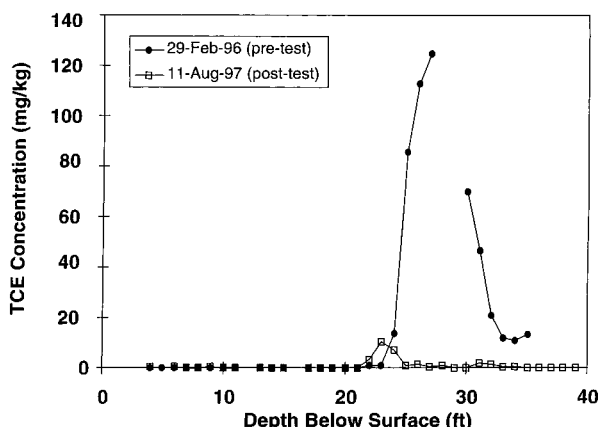


FIGURE 7. Soil TCE concentration profile as a function of depth for location 2A-02.

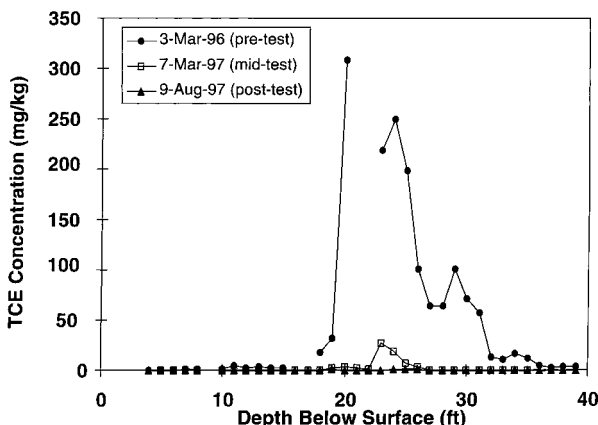


FIGURE 8. Soil TCE concentration profile as a function of depth for location 2A-03.

concentrating between 15 and 30 feet (4.6–9.2 m). Location 2A-03 had the highest TCE concentration and likely contained DNAPL.

The mid-test soil sampling conducted in early March 1997 after six months of operation shows encouraging results. Because of the excessive amounts of TCE found trapped in the carbon socks, this sampling procedure was modified to include samples at all soil strata interfaces, where DNAPL would likely reside, as well as the 30-cm sections. The soil samples near 2A-03, which was located between the 2 ft spaced treatment zones, indicated 95% removal of TCE. Since 2A-03 had the highest pretest TCE levels, the process was

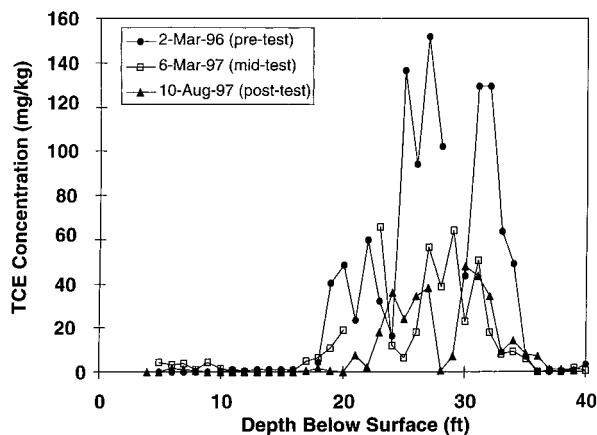


FIGURE 9. Soil TCE concentration profile as a function of depth for location 2A-04.

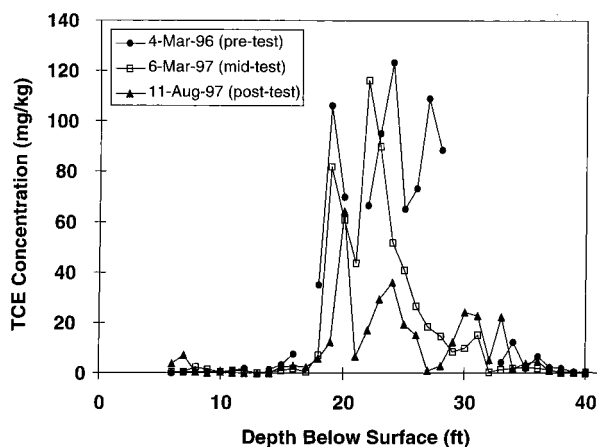


FIGURE 10. Soil TCE concentration profile as a function of depth for location 2A-05.

clearly working, even with the possible presence of DNAPL. Soil samples near 2A-04 showed an average removal of 62%, while the 2A-05 samples showed a 48% removal. The soil samples near 2A-01, which was upstream of a treatment zone, showed very high levels of TCE. Note that this intermediate sampling uncovered the presence of DNAPL at this location.

The post-test samples taken in August 1997 show a continued decline in the soil TCE concentrations. Most impressively, location 2A-03 dropped to a TCE average of less than 0.2 ppm with the highest single sample at 1.5 ppm, resulting in an overall removal of 99.7% for this spot. Location 2A-02 shows excellent clean up with 95% removal and an average soil TCE concentration of less than 1 ppm. Location 2A-01 exhibited only 43% removal (73% if based on mid-test soil samples), but this was expected because the "clean front" of pore water, which had traveled a total distance of 5.6 ft (1.7 m), was just reaching this location by the end of the test.

Locations 2A-04 and 2A-05 had average TCE concentrations above the 5.6 ppm target but show improvement over the mid-test values with 72% and 71% removal, respectively. It is not clear why these two locations were not cleaned as well as 2A-03 since they were also located about 1 ft downstream of a treatment zone. One major difference is that both 2A-02 and 2A-03 were bracketed on both sides with treatment zones, whereas there was none downstream of 2A-04 and 2A-05 except for the cathode, which was 6 ft away. Unlike treatment zones, which could efficiently destroy TCE and also formed a hydraulic barrier (due to the presence of clay), the cathode had uncharacterized dechlorination activity and contained granular materials designed for high hydraulic flow. It was observed during the test that the water

level was somewhat erratic in this area, suggesting significant water movement through some sections. It is possible that some backward movement of TCE in areas of high concentration in this region may be slowing the measured cleanup.

Even though the soil results appear mixed, they may actually show a consistent pattern when viewed in light of whether the sample locations were between two treatment zones and how much purge water had passed through each soil zone. Shown in Table 5, when the contaminated soils were bracketed with treatment zones (2A-02 and 2A-03), significant cleanup occurred. And the extent of cleanup seems to correlate well with the number of pore volumes of water passing through the bracketed soil zone, with less than 3 pore volumes sufficient to remove TCE to significantly below the cleanup target. Based on these field results, the design for the overall cleanup has all of the contaminated soil bracketed with treatment zones, with primarily 5 ft spacing and 2–3 pore volumes. In regions of suspected high DNAPL, some 2 ft spacings will also be used to ensure sufficient flushing between treatment zones for complete TCE destruction.

Volatilization Testing. Because the soil was heated to an average temperature of over 60 °C (>80 °C at the center), there was concern that TCE was being volatilized into the surrounding atmosphere. An EPA-certified flux chamber method (21) was conducted twice during the test, once about halfway into the test and once near the end when the soil temperature was the highest. The flux chamber method is the most commonly used air sampling procedure to determine gaseous emissions from landfills and impoundments. No significant emissions of TCE and its daughter products were detected. The fact that the soil has very low permeability and that the surface is covered with a layer of 1.5 m thick gravel/clay soil may have helped contain the emission.

Treatment Costs. A cost optimization model of the overall treatment process was developed utilizing the cost/performance data obtained from the field test. Three key parameters are used in the design of the Lasagna process: number of electrode rows, number of treatment zones per electrode pair, and the applied electrical potential. Input parameters to the model include soil properties, depth of contamination, cost of emplacing electrodes and treatment zones, required number of purge pore volumes, time constraints to achieve cleanup, and electricity cost (22).

Various cases were run to provide representative cost ranges for applying the technology to clean up TCE contamination in clay. The treatment cost for a one-acre site ranges from \$45 to \$80 per cubic yard depending the depth of contamination (15–45 ft) and remediation time (from 1 to 3 years). For DNAPL cases, the cost goes up with increasing TCE concentration in soil because more purge water volume, hence longer time, will be needed. If the DNAPL migration velocity is assumed to be one-tenth the water migration rate, the cleanup cost would cap at approximately 2500 ppm TCE in soil (about \$100 per cubic yard, requiring 12 pore volumes and six years treatment time). For the Paducah final cleanup, which covers 1/4 acre and reaches 45 ft deep, the treatment cost is estimated to be \$80 per cubic yard for a two year treatment time.

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