Centrifuge Modelling of Contaminant Transport Processes

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ABSTRACT

Over the past decade, research workers have started to investigate problems of subsurface contaminant transport through physical modelling on a geotechnical centrifuge. A major advantage of this apparatus is its ability to model complex natural systems in a controlled laboratory environment In this paper, we discusses the principles and scaling laws related to the centrifugal modelling of contaminant transport, and presents four examples of recent work that has been carried out in this area. The first two of these examples illustrate the use of centrifugal techniques to investigate contaminant transport mechanisms in geologic formations, while the latter two illustrate the use of the centrifuge as a tool for investigating site remediation strategies. The scope of this work serves to demonstrate the contribution that centrifuge modelling techniques can make in the areas of environmental engineering and contaminant hydrology.

KEYWORDS: Centrifuge modelling, geo-environmental engineering, contaminant transport, site remediation

INTRODUCTION

The primary objectives of modelling are simulation and prediction. For example, in the case of sub-surface contaminant transport, modelling can provide valuable information about the spread of contaminants and the effectiveness of various containment and remediation strategies. Thus, modelling can offer much assistance in identifying optimal strategies for waste management.

Predicting the movement of fluids and contaminants in geologic systems is often carried out using theoretical models. In this approach, the processes under examination are simulated by a set of governing equations that are solved by analytical or numerical methods. At present, it is exceedingly difficult to obtain accurate field data that quantitatively describe contaminant transport and fate variables (Huling and Weaver, 1991). Under these circumstances, model validation, a necessary precursor to routine model use, is problematic. For this reason, experimental data obtained under well defined conditions are fundamental for (i) assessing the predictive capabilities of existing theoretical models; and (ii) guiding the formulation, and parameterisation, of improved predictive models.

Over the past decade, research workers have begun to investigate problems of subsurface contaminant transport through physical modelling in a geotechnical centrifuge. A major advantage of this apparatus is its ability to model complex natural systems in a controlled laboratory environment In this paper, we briefly discuss the principles and scaling laws related to the centrifuge modelling of contaminant transport, and present some recent examples of innovative work that has been carried out in this area.

PRINCIPLES OF CENTRIFUGE MODELLING

Geotechnical centrifuge modelling is an experimental method used by engineers to obtain soil stress conditions that are homologous in model and prototype (e.g., Schofield). This is achieved by subjecting a scaled soil model, where all linear dimensions are reduced by a factor *n*, to a centrifugal acceleration of *n* gravities, *ng* (Figure 1). The application of centrifugation in soil science is also well established: Over 20 years ago, Alemi et al. (1976) demonstrated the experimental determination of transport parameters in soil cores using centrifugal techniques. More recently, research work in the field of environmental engineering has made use of the method to investigate the behaviour of gravity-driven flow phenomena under realistic, but well controlled, boundary conditions (e.g., Cooke and Mitchell, 1991; Hensley and Savvidou, 1993).

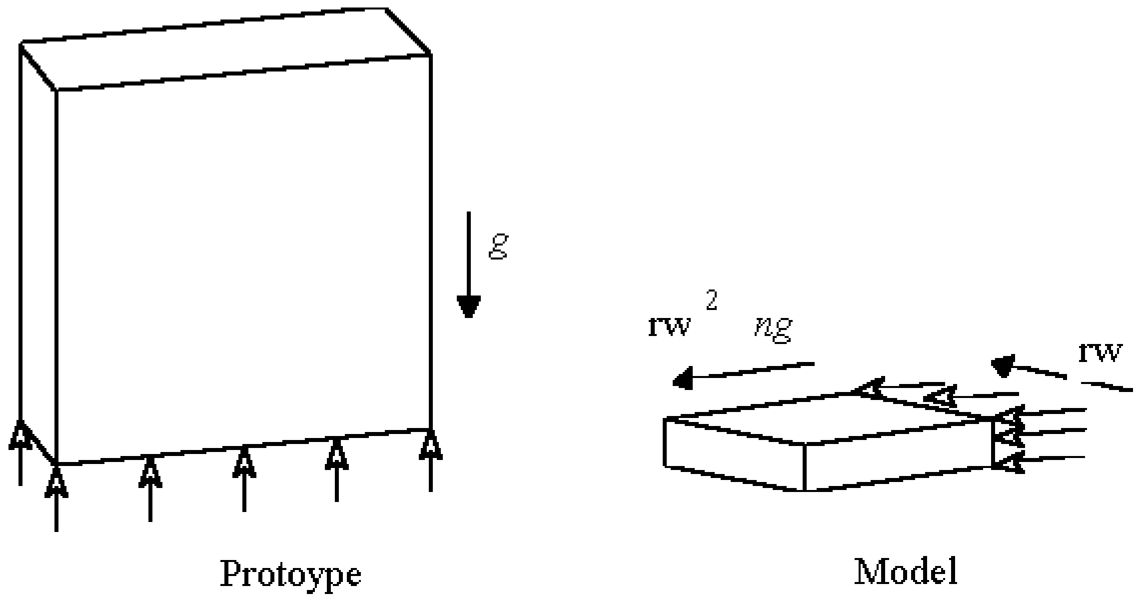


Figure 1. Gravity effects in model and prototype (after Schofield, 1980).

Scaling relationships for flow phenomena in the centrifuge have been developed by numerous authors (e.g., Arulanandan et al., 1988). Culligan-Hensley and Savvidou (1995) present a detailed overview of these relationships. For a reduced-scale centrifuge model test conducted using prototype soils and liquids, the scaling ratios given in Table 1 are either self-evident or well established. It is assumed that these ratios are also applicable to the centrifuge model studies presented below.

Table 1. Centrifuge scaling relationships for a scale factor of *n*. The same soil and fluids are used in model and prototype.

Parameter	Prototype /model ratio
Gravity, *g*	$1/n$
Macroscopic length, L	n
Microscopic length, d	1
Pore fluid velocity, u	$1/n$
Time	n^2
Fluid pressure, p	1
Hydraulic conductivity, K	$1/n$
Soil intrinsic permeability, k	1
Soil porosity, ϕ	1
Fluid density, ρ	1
Fluid viscosity, μ	1
Fluid interfacial tension, σ	1
For diffusion dominated spreading:	
Dissolved contaminant concentration	1

CENTRIFUGE MODEL STUDIES

Four centrifuge model studies are presented here. In the first two of these studies, the aim was to examine physical mechanisms or contaminant transport, while in the latter two, the focus was on investigating techniques for hazardous waste-site remediation.

Solute Transport in Heterogeneous Soil

Background

Many natural soils exhibit physical heterogeneity (e.g., heterogeneity in hydraulic conductivity) that gives rise to an "apparent" time-dependent sorption of chemicals during contaminant transport.

The transport of a non-reactive contaminant (solute) undergoing "apparent" time-dependent sorption (usually referred to as "physical non-equilibrium sorption") is often described mathematically by a two-region soil model (TRM) (Coats and Smith, 1964). In the usual onceptualisation of the TRM, the porous medium is divided into "mobile" and "immobile" regions. Advective and dispersive solute transport occurs in the first, mobile soil region, while fluid in the second, immobile region is assumed to be stagnant. Thus, in conceptualisation, the immobile region acts as a source (or sink) for solute in the mobile region: with the rate of mass transfer typically taken to be proportional to the concentration difference between the regions. Although numerous research workers have shown that the TRM can adequately describe contaminant transport with physical, non-equilibrium sorption (see, for example, De Smedt and Wierenga, 1979), the model's application is, in practice, limited by the difficulties in determining the model parameters. Under these circumstances, physical modelling may be of practical use as a complimentary technique with mathematical modelling. Here, we describe experiments undertaken by Li et al. (1993, 1994a, b) to investigate the potential for centrifuge modelling of contaminant transport with physical non-equilibrium sorption. The work was conducted using the geotechnical centrifuge facility in the Department of Civil Engineering at the University of Western Australia. A description of this facility is given by Fahey et al. (1990).

Approach

Six model tests were performed that incorporated the principle of "modelling of models" (where centrifuge models of different scales are tested at appropriate accelerations, such that they correspond to the same prototype). The model tests were carried out using an artificially constructed two-region soil structure. Polyethylene porous cylinders (PPC), having a hydraulic conductivity of 4.2×10^{-5} m/s, were used to simulate one of the soil regions. These cylinders were then interspersed uniformly in a less permeable, homogeneous soil having a hydraulic conductivity of 3.5×10^{-6} m/s. Sodium chloride was used as the non-reactive contaminant. Clearly this soil structure cannot be considered as being composed of mobile and immobile regions, since the majority of the flow takes place in the less permeable "mobile" region.

Two models were used during the test series, Model 1 and Model 2. Both models were constructed in a plastic-lined, aluminum column of 140 mm diameter. For Model 1, the soil column was 240 mm long, which was twice that used for Model 2. Model 1 was tested under three different accelerations ($g_1 = 10g$, $g_2 = 30g$, $g_3 = 50g$), while Model 2 was tested under three corresponding accelerations ($g'_1 = 20g$, $g'_2 = 60g$, $g'_3 = 100g$). Figure 2 shows the configuration of each model. A conductivity probe was used to measure the concentration of contaminated effluent as a function of time at the base of the model.

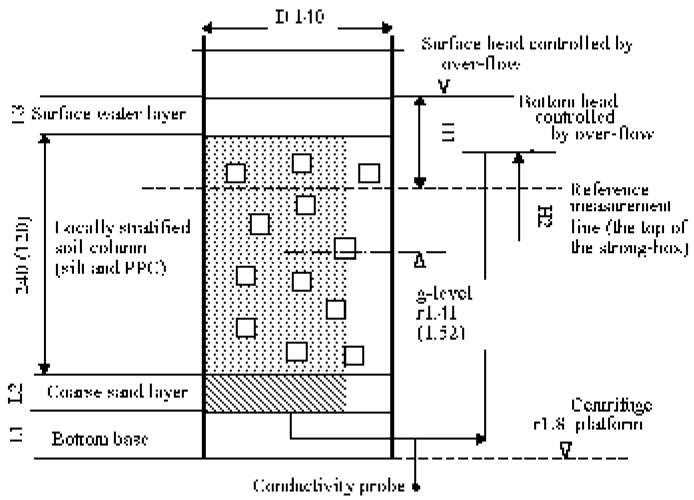


Figure 2. Form of centrifuge model used to investigate solute transport in heterogeneous soil. The values in parentheses are for Model 2. L1 = 120 mm; L2 = 45 mm; L3 = 13 mm (6.5 mm); H1 = 43 mm (-77 mm, i.e., below the reference measurement line); H2 = 7 mm (-95 mm) (after Li et al., 1994a).

Results and Discussion

In heterogeneous soils described by the TRM, inter-region mass transfer is mainly affected by two factors, namely, the local flow variations and inter-region diffusion (Li, 1993). When the fluid flow rate is low, the dominant factor is the local flow conditions. Under these circumstances, dimensional analysis shows that the inter-region mass transfer rate should scale correctly during centrifuge modelling provided that contaminant spreading is diffusion dominated (Li et al., 1993).

During all centrifuge tests, a steady, but low, flow rate was maintained through the two-region soil column. A maximum grain-Peclet number of 0.8 was recorded for the test series. This ensured that longitudinal contaminant spreading was predominantly by diffusion (Bear, 1972). By using the relationships presented in Table 1, the scaled data from Model 2 can be plotted with the data from Model 1. Figure 3 compares selected scaled data from Model 2 with similar data from Model 1. The good agreement between these results demonstrates the feasibility of modelling Model 1 by Model 2. This suggests that the mechanisms governing contaminant transport, including inter-region mass transfer, were similar in both models. This, in turn, indicates that modelling physical non-equilibrium transport processes with relatively slow flow may be feasible using the geotechnical centrifuge.

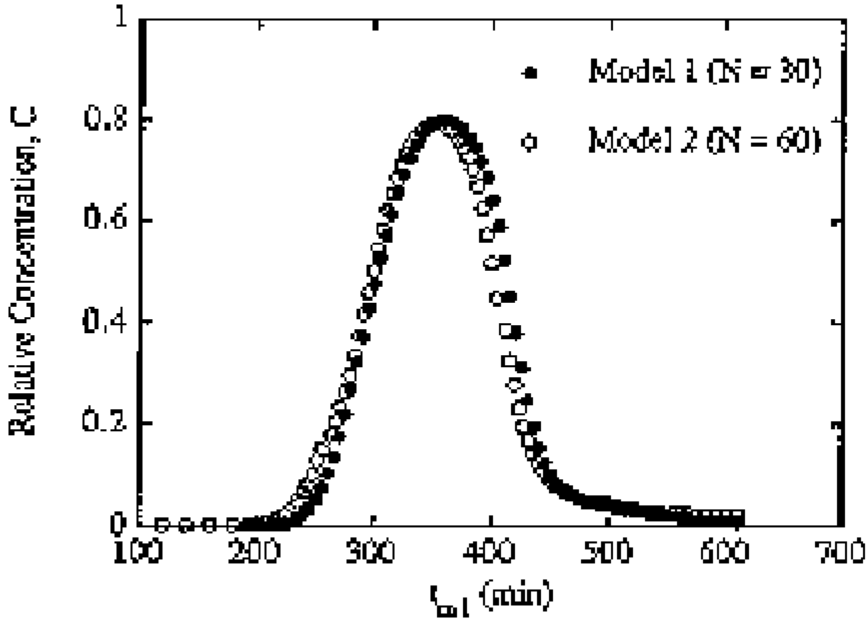


Figure 3. Comparison of data from "modelling of models" tests. t_{m1} is the un-scaled time for Model 1 (after Li et al., 1994a).

Summary

The work presented here demonstrated that centrifuge modelling of physical non-equilibrium sorption in a two-region soil structure under low-flow conditions is possible. Because the scaling depended on the flow variations being the cause of apparent nonequilibrium, the mobile/immobile region conceptualisation usually attributed to the TRM is not applicable in this case. The results show that the geotechnical centrifuge might provide a useful technique for studying non-equilibrium contaminant transport in soils, particularly in soils where there might be stress-dependency of physical heterogeneity.

NAPL Infiltration in the Vadose Zone

Background

Non-aqueous phase liquids (NAPLs), such as gasoline, chlorinated solvents and PCB (polychlorinated biphenyl) oils, are a common cause of groundwater contamination in many industrialised countries. These liquids exist as a separate fluid phase in the subsurface, yet typically have solubilities orders of magnitude greater than drinking water standards (e.g., Kueper et al., 1992). Predicting the fate of these chemicals in the subsurface is a challenging problem that needs to be addressed at many sites, before an effective remediation result can be achieved.

Geotechnical centrifuge modelling is a technique that has proven useful in the study of stable, nonaqueous phase liquid (NAPL) infiltration in unsaturated media (e.g., Illangasekare et al., 1991). Under certain conditions, however, NAPL fronts infiltrating unsaturated media can become unstable and break into fingers. These can move vertically to the water table, by passing a large proportion of the vadose zone (e.g., Glass et al., 1988; Barry and Li, 1994). In general, unstable fingering occurs during NAPL infiltration into regions where the hydraulic conductivity increases in the direction of positive flux (i.e., in the vertical direction). NAPL transport under these circumstances is extremely non-uniform. As a result, contaminant loading at the water table will be quite different than if transport is assumed to be one-dimensional and stable.

In this section, we present recent work undertaken by Banno (1996) to investigate the feasibility of modelling unstable NAPL infiltration in the vadose zone using a geotechnical centrifuge. This work was conducted using the geotechnical centrifuge facility in the Department of Civil and Environmental Engineering at Massachusetts Institute of Technology (MIT); the reader is referred to Pahwa (1987) for a detailed description of this machine.

Gravity-driven Wetting Front Instability

Because a nonaqueous phase liquid will preferentially wet a solid with respect to air, the infiltration of NAPL in the vadose zone involves a wetting front displacement. When the invading fluid is wetting, the width and separation of fingers that form due to gravity-driven instability are macroscopic: This is because wetting fluids exhibit spreading behaviour during propagation into a porous medium (Homsy, 1987). Thus, if finger scaling during centrifuge modelling is feasible, the width and separation of unstable fingers in a centrifuge model should be n times smaller than those in the prototype, while the velocity of the fingers should be n times higher (cf., Table 1). This observation has been confirmed through scaling analysis by Culligan et al. (1996), and applies equally to the case where the invading fluid is water.

Approach and Results

The work involved a set of model experiments in a glass fronted box. In all tests, the light nonaqueous phase liquid (LNAPL) Soltrol 220, ponded at a constant depth on the model surface, was allowed to infiltrate into an initially dry medium comprising two sand layers. The experimental set-up is shown in Figure 4. Because the hydraulic conductivity of the upper sand layer was lower than that of the lower layer, fingers were expected to form as the infiltrating front passed the interface between the two layers. Experiments were conducted at g-levels ranging between $1g$ (i.e., at the prototype scale) and $11g$. Video recordings were used to determine the average finger width, finger velocity and finger separation during each experiment.

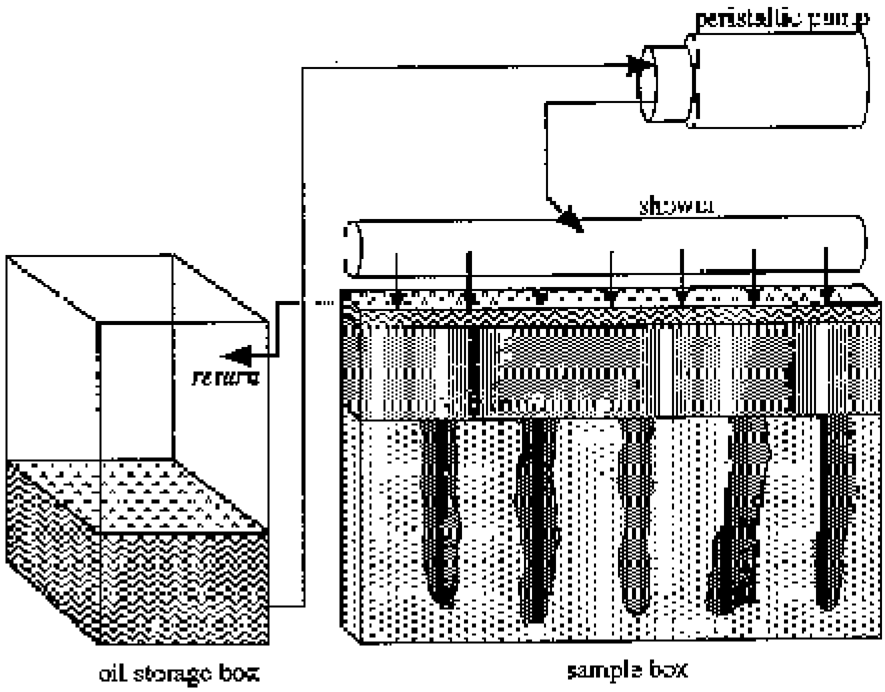
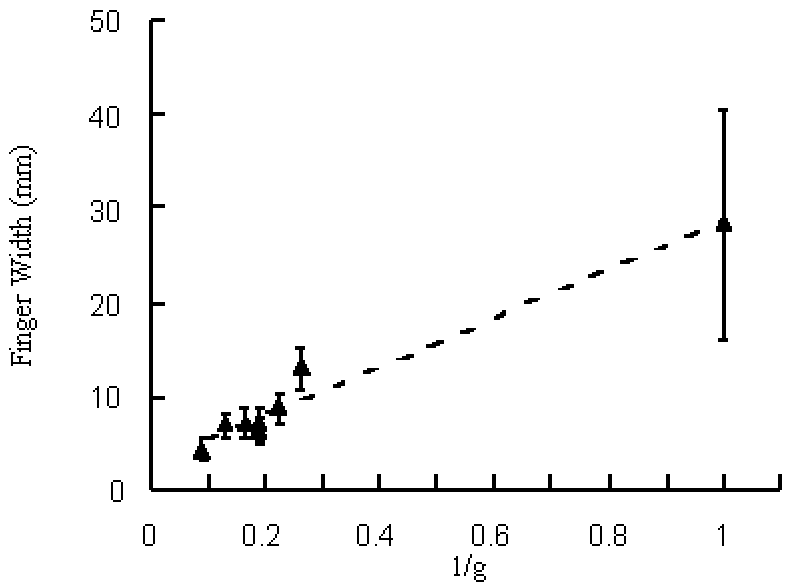


Figure 4. Form of centrifuge model used to investigate NAPL infiltration in the vadose zone (from Banno, 1996).

Figure 5 presents data on the average finger width, separation and velocity for each test (Banno, 1996). These data shown that the finger width, separation and velocity vary linearly with *g*-level, and in accordance with the scaling relationships presented in Table 1.

Thus, it can be concluded that scaling of gravity-driven NAPL fingering in the vadose zone is possible using the geotechnical centrifuge.



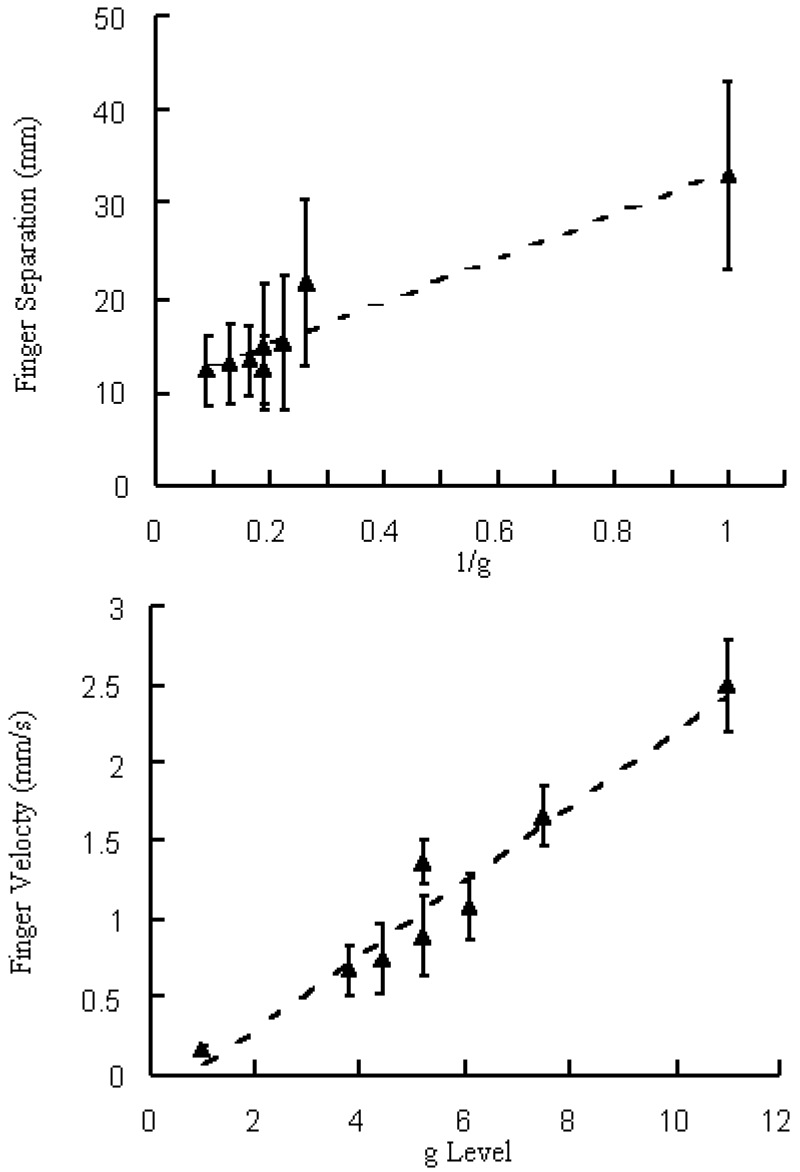


Figure 5. Variation of finger properties with *g*-level (after Banno, 1996). (a) Finger width versus 1/*g*; (b) finger separation versus 1/*g*; (c) finger velocity versus *g*. The solid symbols represent the average observed value. The solid lines represent the range of observed values. The dashed line is a linear regression analysis performed using the data.

Summary

Results from this work demonstrated the feasibility of using centrifugal techniques to study unstable NAPL infiltration in dry soil. Data from a preliminary test conducted to investigate unstable infiltration in a soil model accommodating a representative moisture content profile above the water table are also promising; they suggest that a geotechnical centrifuge offers a useful tool for studying unstable NAPL transport under realistic field boundary conditions.

LNAPL Behaviour under Hydraulic Flushing

Background

Over the past decade, the technological response to demands for clean-up at many hazardous waste sites has exclusively been the application of so-called "pump-and-treat" technology. In essence, pump-and-treat technology attempts to "flush out" the contaminants by physical displacement to an extraction well, where the recovered effluent can be treated with established above ground technologies.

The majority of the most frequently detected contaminants at hazardous waste sites in the United States are non-aqueous phase liquids (USEPA, 1992). When the extraction of these chemicals is undertaken by hydraulic flushing (water flooding), discontinuous "blobs" of NAPL can become entrapped in the porous medium behind the wetting (water) front. The entrapment of NAPLs under hydraulic flushing is pertinent to many aspects of site remediation. Entrapment mechanisms determine what proportion of NAPL can be removed using traditional pump-and-treat technology and, consequently, how much NAPL remains for possible recovery by alternative methods, such as bioremediation, *in-situ* air sparging or surfactant flushing.

This section describes a research program under taken by Ratnam et al. (1996a, b), who made use of centrifugal modelling techniques to investigate LNAPL entrapment under hydraulic flushing. The work was performed using the centrifuge facility in the Department of Civil and Environmental Engineering at MIT.

Mechanisms for NAPL entrapment

When a wetting fluid (such as water) displaces a nonwetting fluid (such as oil) in a porous medium, a proportion of the nonwetting fluid will remain entrapped in the medium pores once the wetting front has passed. In general, the nonwetting fluid is trapped in the form of disconnected "blobs" or "ganglia," whose complicated morphology and distribution is dependent upon (i) the geometry of the pore space; (ii) fluid-fluid properties such as interfacial tension, density contrast, and viscosity ratio; (iii) the fluid-solid interfacial properties effecting wetting behaviour; (iv) the applied pressure gradient; and (v) gravity (Morrow and Songkran, 1981).

During stable fluid displacement, nonwetting phase (NAPL) entrapment is caused by capillary forces and overcome by viscous pressure forces and/or buoyancy forces (Mayer and Miller, 1992). The ration of viscous pressure to capillary forces and gravity to capillary forces are usually expressed as dimensionless groups, known as the capillary number (C_a) and the Bond number (B_o), respectively.

$$C_a = \frac{u\mu}{\sigma} \qquad B_o = \frac{\Delta\rho g R^2}{\sigma} \qquad (1)$$

where u is the displacing (Darcy) fluid velocity, μ is the displacing fluid viscosity, σ is the interfacial tension, $\Delta\rho$ is the fluid density contrast, g is the acceleration due to gravity, and R is the average particle radius of the porous medium.

Above a certain upper, critical capillary or Bond number, no capillary trapping occurs, and the residual nonwetting phase saturation is close to zero. Conversely, below a certain lower, critical capillary or Bond number, the trapping is dominated by capillary forces, and the residual nonwetting phase saturation reaches a constant maximum value.

Between these two extremes, the residual nonwetting phase saturation depends upon the combined effects of capillary, viscous pressure, and gravity forces.

The mapping of these three domains in the plane C_a - B_o, constitutes the "phase-diagram" for a given system. The aim of this work was to establish experimental phase-diagrams for several uniform porous media systems, under vertical displacement conditions.

Approach

For the particular case of nonwetting phase entrapment, centrifuge modelling offers a simple means of varying the Bond number characterizing a particular system: this is achieved by simply varying the body force (g-level) acting upon that system (refer to Equation 1). No other experimental technique can offer the same versatility as centrifuge modelling in this respect.

Figure 6 shows a section through the model that was used in the study. All experimental models were constructed from uniform silica sand fully saturated with the aliphatic oil Soltrol-220. During each experiment, the nonwetting phase fluid (Soltrol-220) was displaced upward at a constant velocity by water, which was acting as the wetting phase fluid. The wetting phase displacement velocity was controlled by a constant rate displacement pump. The degree of entrapped nonwetting phase remaining after a single passage (one pore volume) of the wetting front was determined from appropriate pre- and post-test weight and volumetric measurements.

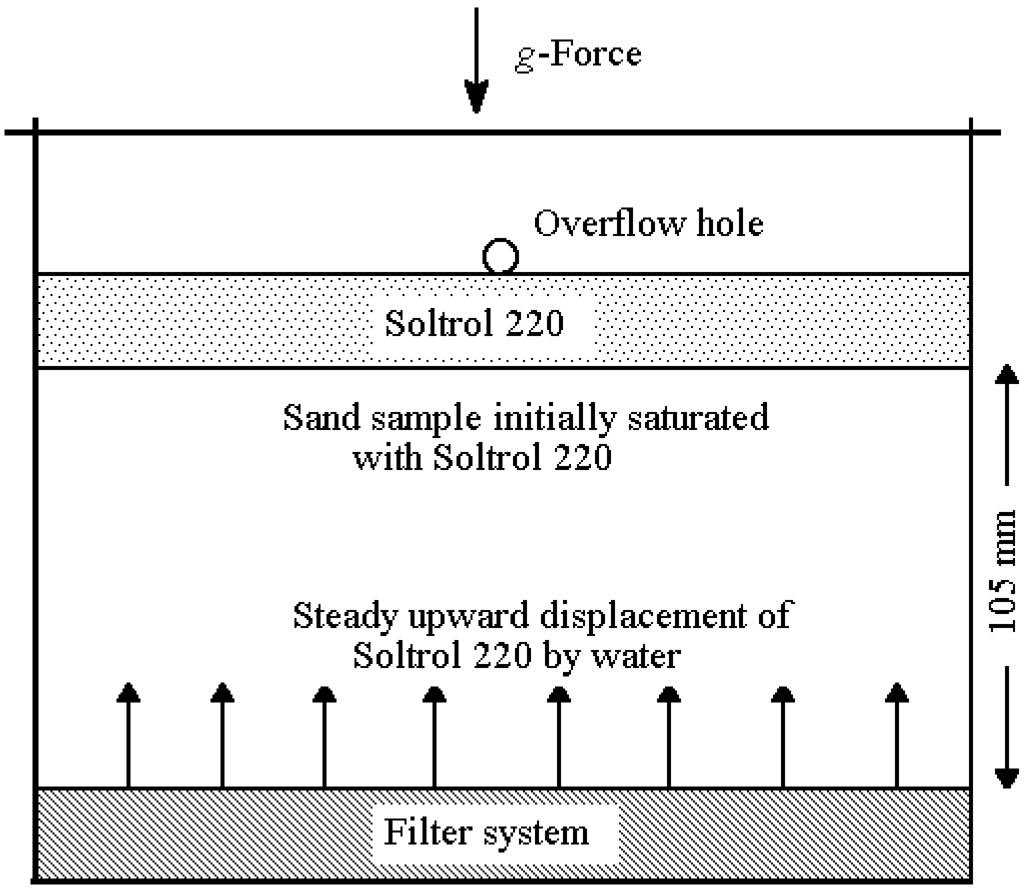


Figure 6. Form of centrifuge model used to investigate LNAPL behaviour under hydraulic flushing (after Ratnam et al., 1996).

The effective sum of the capillary and Bond numbers was varied between tests, by changing the wetting phase displacement velocity through the model and/or the gravitational force acting on the model.

Four porous medium systems were used during the investigation, three comprising silica sands and a fourth comprising glass beads (Table 2). All sands modelled a uniform, high permeability aquifer system, typical of formations where pump-and-treat technologies have proven to be most effective (NRC, 1994).

Table 2. Properties of silica sands and glass beads (after Ratnam et al., 1996a).

Soil Type	D_{50} (mm)	Uniformity Coefficient	Specific Gravity	Intrinsic Perm. (m²
40P	0.35	1.6	2.66	6×10^{-11}
90P	0.17	1.6	2.66	2×10^{-11}
100P	0.17	1.8	2.66	2×10^{-11}
Glass Beads	0.35	--	2.46	--

Results and Discussion

Thirty vertical displacement tests were conducted using the various porous media described. Table 3 presents typical data from a selection of these tests. For each porous medium system, the variation in the sample void ratio was kept to a minimum between experiments.

Table 3. Experimental results for LNAPL entrapment in Type 40P sand (after Ratnam et al., 1996b).

g-level	$C_a \times 10^6$	$B_o \times 10^2$	Trapped residual S (%)
1.0	3.11	0.24	9.35
1.0	3.11	0.24	9.30
7.25	3.95	1.77	4.38
14.02	3.70	3.42	3.35
35.83	4.19	8.7	2.90
37.18	3.46	9.1	2.84
57.98	3.23	14.1	2.36

In the case of the upward displacement of an LNAPL by water, viscous pressure and buoyancy forces act in the same direction, and are thus additive in overcoming the mechanism of capillary entrapment. When this occurs, it is possible to combine the individual effects of the capillary and the Bond number in the following dimensionless group (Morrow and Songkran, 1981).

$$N_{CB} = C_a + k_{wf} A B_o \qquad \textbf{(2)}$$

where k_{wf} is the relative permeability of the wetting phase at the flood front, and A is a constant presumed to link the intrinsic permeability of a porous medium to its average particle radius, viz.,

$$k = AR^2 \qquad \textbf{(3)}$$

Based on previous work (Naar et al., 1962), an average relative permeability k_{wf} = 0.63 was assumed at residual LNAPL saturation for all sands, while the A value for each sand type was computed from the data presented in Table 3.

Figure 7 presents the percentage of residual versus N_{CB} for all vertical displacement tests in sand. Note that the results have been normalised with respect to the maximum observed residual LNAPL, S_m(%). The data clearly show the influence of buoyancy and viscous pressure forces on capillary entrapment. At higher values of N_{CB}, the supplemental pressure provided by viscous pressure and/or buoyancy acts to reduce the proportion of LNAPL remaining entrapped in the sand pores once the wetting front has passed.

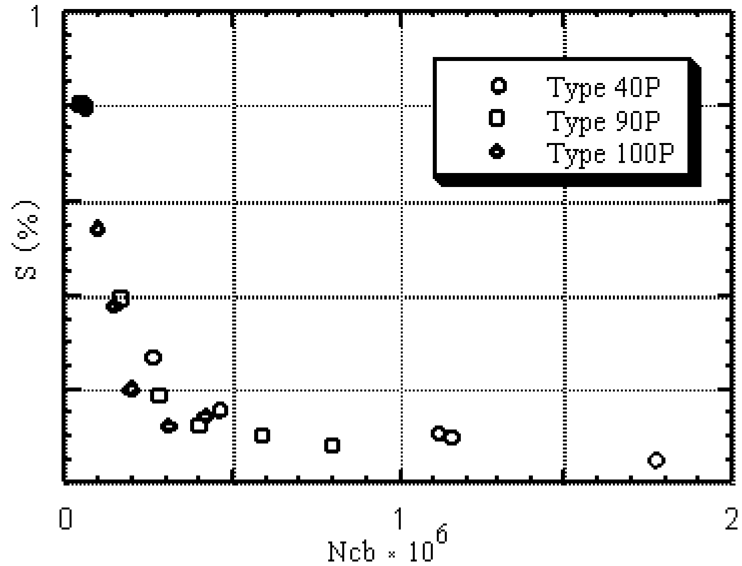


Figure 7. Normalised trapped LNAPL residual saturation versus N_{CB}.

The construction of phase-diagrams, such as that given in Figure 7, is not original and has been reported elsewhere in the literature (e.g., Lenormand et al., 1988). However, the use of centrifuge modelling as an experimental tool to conduct this form of investigation is new, and the work presented here demonstrates the potential offered by centrifuge modelling in this area.

Summary

The behaviour of a light nonaqueous phase liquid under vertical hydraulic flushing was investigated over a wide range of conditions using centrifugal modelling techniques. The work clearly demonstrated the influence of buoyancy and viscous pressure forces on capillary entrapment. Data from experiments conducted in different silica sands were used to construct a phase-diagram describing the percentage of LNAPL trapped in contaminated soil following one pore-volume injection of water, as a function of a combined capillary and Bond number (N_{CB}). This form of diagram may be used to predict the amount of LNAPL trapped behind the passage of a wetting front for any value of N_{CB}.

The data presented here indicate the potential of the geotechnical centrifuge as an experimental tool in this area. This work is currently being extended to investigate nonwetting phase entrapment under horizontal displacement conditions, where the effects of the capillary and Bond number are no longer additive, and in pre-wet media, where the initial antecedent moisture content will also influence capillary entrapment.

Electrokinetic Clean-up of Contaminated Land

Background

Electrokinetic clean-up is a technique that can be used to remediate low permeability soils.

This can be achieved by application of a direct current across the contaminated soil using electrodes that are inserted into the ground. The main processes governing the migration of chemicals within an electrical field are ionic migration and electro-osmosis (Mitchell, 1993). Ionic migration is the movement of an ionic species towards the oppositely charged electrode. Electroosmosis is a phenomenon caused by the diffuse double layer present at the particle-fluid interface. The mobile ions within this layer migrate towards the oppositely charged electrode (in clays, generally towards the cathode) producing a drag force on the surrounding fluid and thereby inducing an electro-osmotic flow. This processes is effected by the inter-particle spacing in a soil, which, in turn, will be influenced by the soil stress level.

The work reported here was undertaken by Penn (1997) and Penn and Savvidou (1996) to investigate the potential for centrifuge modelling of electrokinetic remediation of clay soils. The tests were performed using the balanced arm centrifuge facility in the Cambridge University Engineering Department. This facility is described by Schofield (1980).

Approach and Results

Five centrifuge tests were performed in cylindrical models of kaolin clay mixed and saturated with copper sulphate solution. All tests were conducted at 50*g*. Figure 8 shows a schematic diagram of the equipment used. A constant, direct-current electrical potential was applied across the clay using rectangular platinum electrodes. Standpipes were used to maintain a constant fluid head in the electrode wells. Contaminants that migrated to the electrodes under the processes of ionic migration and electroosmosis were removed by flushing the wells with deionised water held in the model reservoirs. The voltage field, temperature and pore water pressures within each model were continually monitored throughout each test. Post-test sampling and chemical analysis were used to determine the final concentration of copper and sulphate ions in the soil pore fluid.

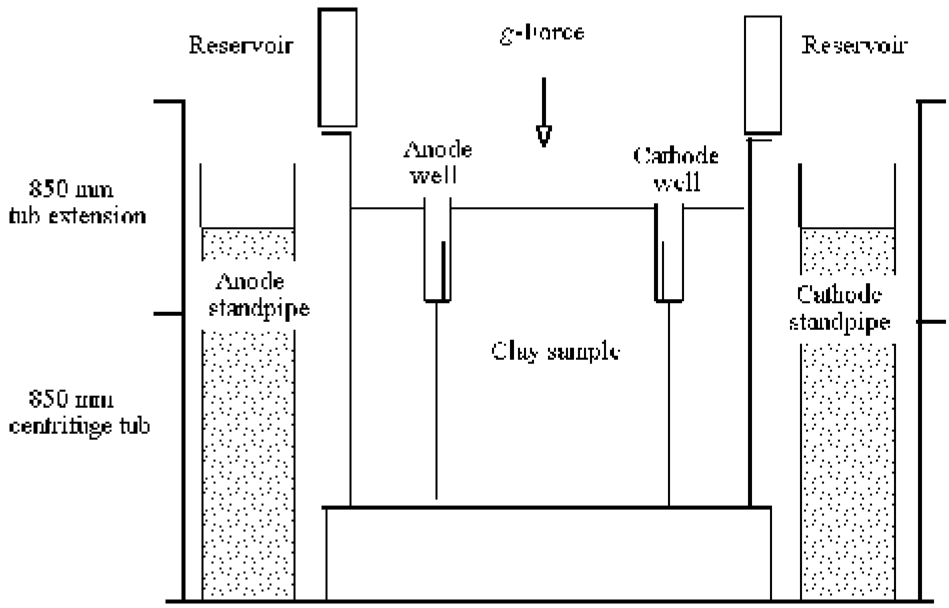
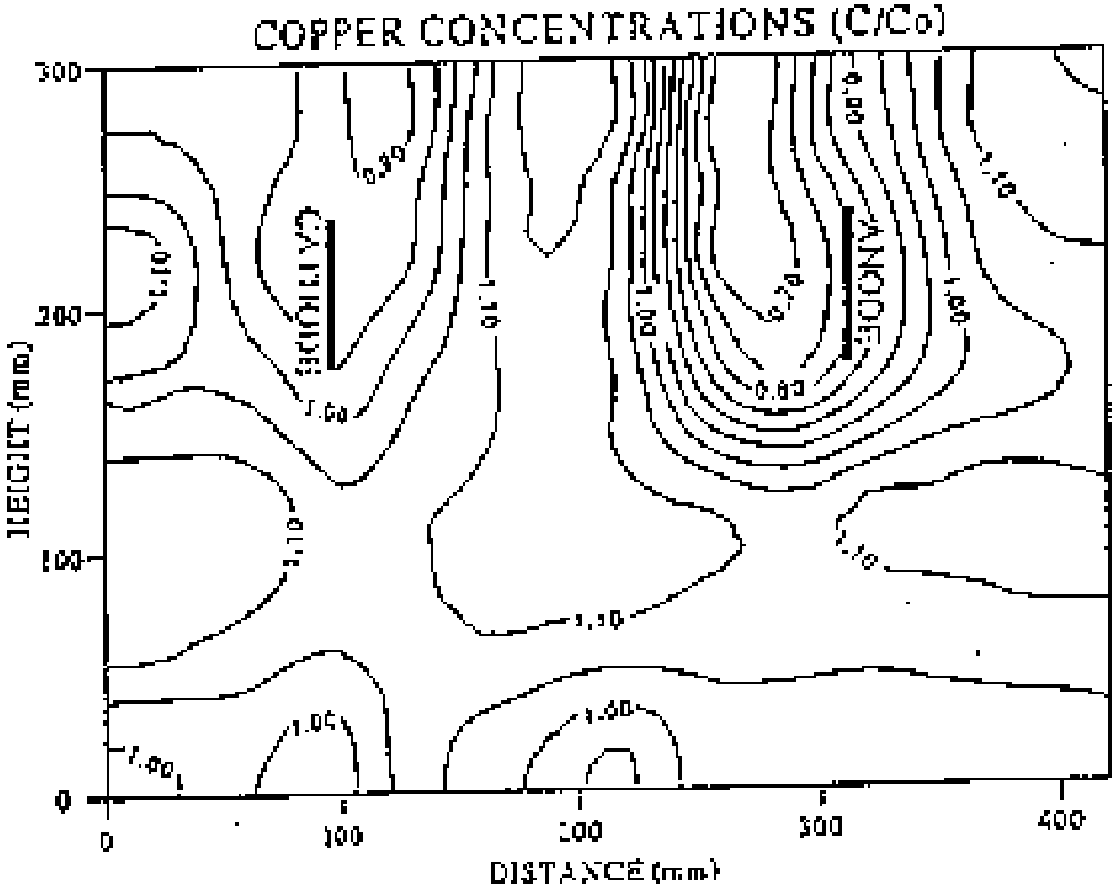


Figure 8. Form of centrifuge model used to investigate electrokinetic clean-up of contaminated soil (after Penn and Savvidou, 1996).

Figure 9 presents the initial and final voltage fields at a cross section through one of the models. The data show that there was a distortion in the initially uniform voltage field during the course of the experiment, indicating a change in soil electrical conductivity with time. In particular, a high electrical resistance was developed in the areas close to the electrodes, especially the cathode. This was caused by the formation of moving acid and alkali fronts in the model as a result of electrolysis reactions occurring in the deionised water (Acar et al., 1990). Copper ions, moving towards the cathode, would have met the alkali front (formed by hydroxyl ions produced at the cathode) as it migrated towards the anode. This intersection would have caused the copper ions to transform into copper hydroxide, a non-conductive precipitate. In addition, electroosmosis around both the anode and the cathode, would have drawn the low conductivity de-ionised water into the surrounding soil, again creating zones of electrical resistance.



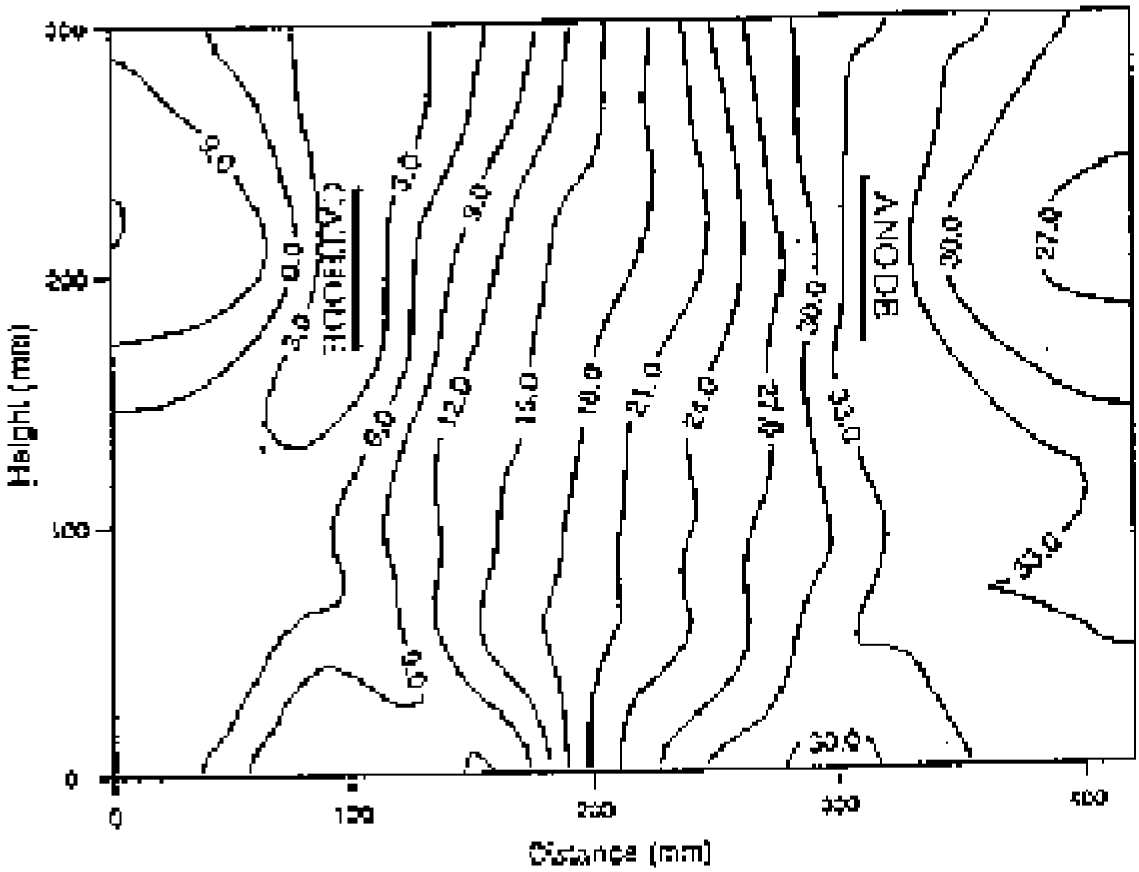
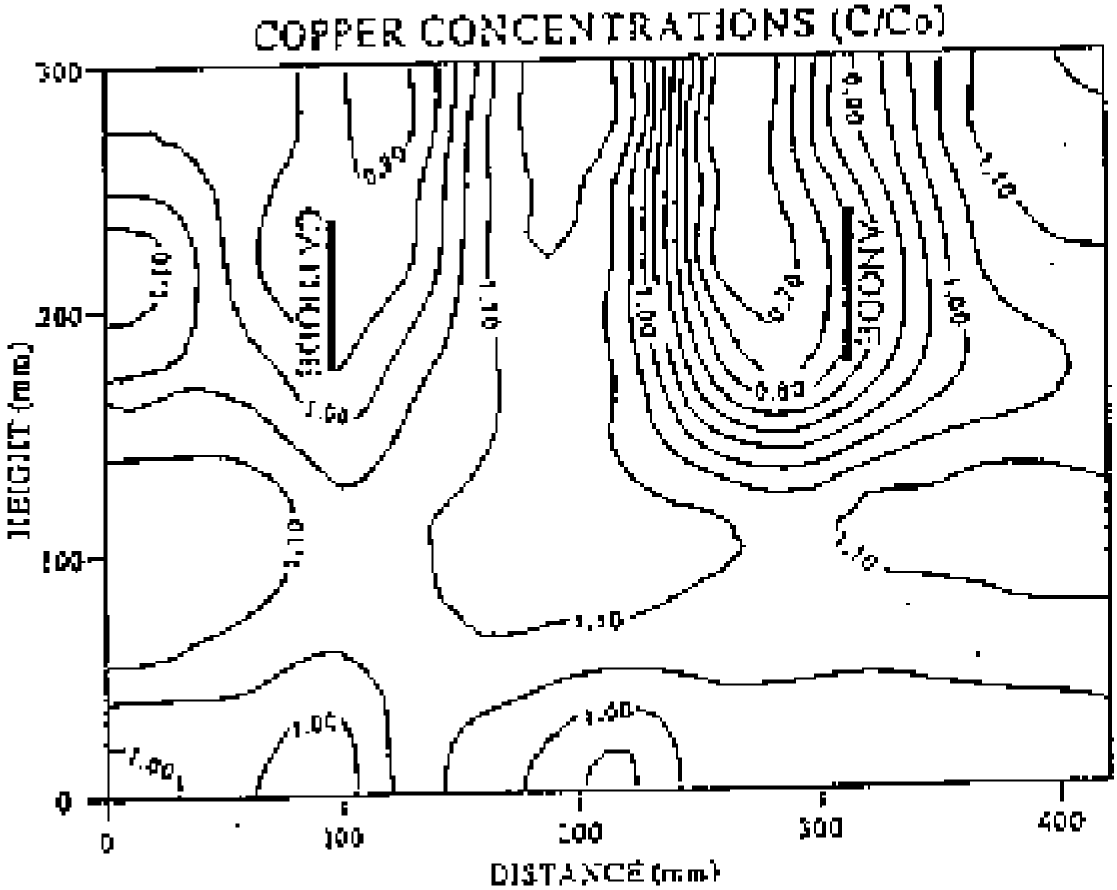


Figure 9. Voltage field (from Penn and Savvidou, 1996); (top figure) initial; (bottom figure) at 22 hours.

It was observed that the areas of high soil resistance were developed shortly after the application of the voltage field. The creation of these areas resulted in a reduction in the current driven through the sample, and an increase in the soil temperature.

Final copper and sulphate concentration for a vertical cross-section through the same model are given in Figure 10. These data show some reduction in ion concentration around the electrodes. However, the results highlight the fact that electrokinetic remediation is susceptible to small areas of high resistance between the electrodes, that limit the process by reducing the driving gradient across the majority of the contaminated soil.


COPPER CONCENTRATIONS (C/Co)
HEIGHT (mm)
DISTANCE (mm)
CATHODE
ANODE

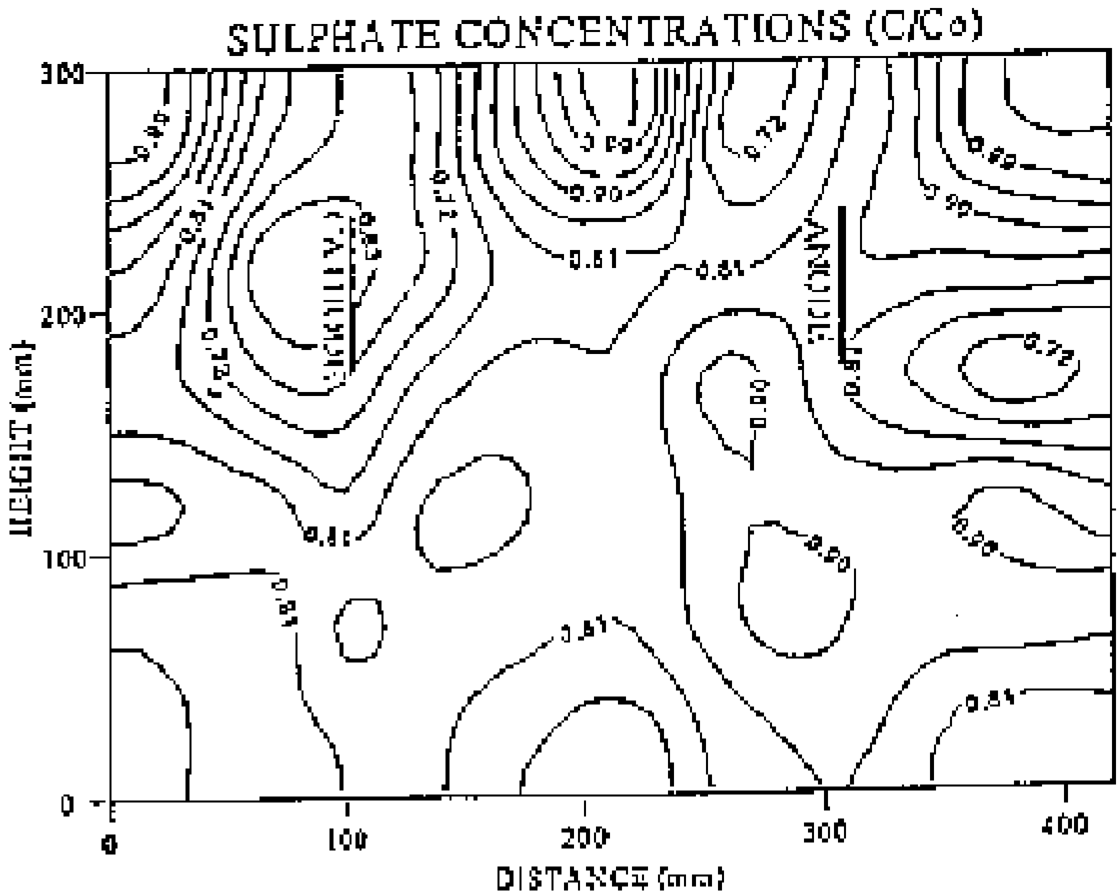

SULPHATE CONCENTRATIONS (C/Co)
HEIGHT (mm)
DISTANCE (mm)
CATHODE
ANODE

Figure 10. Final copper and sulphate contaminant concentrations. (From Penn and Savvidou, 1996).

Summary

This work demonstrated that the migration of an alkali front through contaminated soil undergoing electrokinetic remediation can lead to the formation of precipitates that are detrimental to the remediation process. In addition, the success of the test series suggests that the geotechnical centrifuge might offer a new method of investigating the efficiency of realistic electrokinetic configurations, especially in compressible soils (such as clays) where soil stress levels often influence the material properties controlling this phenomenon.

CONCLUSIONS

In this paper, we discussed the use of geotechnical centrifuge modelling as a tool to investigate problems of subsurface contaminant transport. A major advantage of this apparatus is its ability to model in accelerated time frames, using small scale models, complex natural systems in a controlled laboratory environment. Four examples of recent work that have been carried out in this area were presented. The first two of these examples concerned the use of centrifugal techniques to investigate contaminant transport mechanisms in geologic formations: the latter two illustrated the use of the centrifuge as a tool for investigating site remediation strategies. The scope of this work serves to demonstrate the potential that centrifuge modelling offers as an experimental tool in the areas of environmental engineering and contaminant hydrology.

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REFERENCES

1. Acar, Y.B., Haned, J., Gale, R.J. and Putnam, G. 1990. Bulletin of Transportation Research, Record No. 1288: 23-34.
2. Alemi, M.H., Nielsen, D.R., and Biggar, J.W. 1976. Determining the hydraulic conductivity of soil cores by centrifugation. Soil Sci. Soc. of Am. J., **40**(2): 212-218.
3. Arulanandan, K., Thompson, P.Y., Kutter, B.L., Meegoda, N.J., Muraleetharan, K.K., and Yogachandran, C. 1988. Centrifuge modeling of transport processes for pollutants in soils, J. Geotech. Eng., 114(2), 185-205.
4. Barry, D.A., and Li, L. 1994. Physical basis of nonequilibrium solute transport in soil. *In* Fifteenth International Congress of Soil Science, Transactions. International Society of Soil Science and Mexican Society of Soil Science, Acapulco, Mexico, **2a**: 86-105.
5. Bear, J. (1972). *Dynamics of Fluids in Porous Media*. American Elsevier, New York.
6. Banno, K. 1996. Centrifuge modeling of immiscible fingering in porous media. Thesis, Department of Civil and Environmental Engineering, Massachusetts Institute of Technology, Cambridge, MA., 1996.

7. Coats, K.H. and Smith, B.D. 1964. Dead-end pore volume and dispersion in porous media. Soc. Petrol. Eng. J., **4**(3), 73-84.
8. Cooke, B., and Mitchell, R.J. 1991. Physical modelling of a dissolved contaminant in an unsaturated sand. Can. Geotech. J., **28**(6): 829-833.
9. Culligan, P.J., Barry, D.A., and Parlange, J.-Y. 1996. Scaling unstable infiltration in the vadose zone. Accepted for publication in Canadian Geotechnical Journal (Sept. 1996).
10. Culligan-Hensley, P.J., and Savvidou, C. 1995. Environmental geomechanics and transport processes. *In* Geotechnical centrifuge technology. *Edited by* R.N. Taylor, Blackie Academic, London, pp. 196 - 263.
11. Fahey, M., Finnie, I., Hensley, P.J., Jewell, R.J., Randolph, M.F., Stewart, D.P., Stone, K.J.L., Toh, S.H. and Windsor, C.S. 1990. Geotechnical centrifuge modelling at the University of Western Australia, Australian Geomechanics, **19**: 33-49.
12. Glass, R.J., Steenhuis, T.S., and Parlange, J.-Y. 1988. Wetting front instability as a rapid and far-reaching hydrologic process in the vadose zone. J. of Contam. Hydrol. **3** (2-4): 207-226.
13. Hensley, P.J., and Savvidou, C. 1993. Modelling coupled heat and contaminant transport in groundwater. International Journal for Numerical and Analytical Methods in Geomechanics, **17**(7):493-527.
14. Homsy, G.M. 1987. Viscous fingering in porous media, Ann. Rev. Fluid Mech., **19**: 271-311.
15. Huling, S. G., and Weaver, J. W. 1991. Dense nonaqueous phase liquids, EPA Groundwater Issue, *Center for Environmental Research Information*, Rep. No. EPA/540/4-91/002, 21 pp.
16. Illangasekare, T.H., Znidarcic, D., Al-Sheridda, M., and Reible, D.D. 1991. Multiphase flow in porous media. *In* Centrifuge 91, *Edited by* H.-Y. Ko and F.G. McLean. Balkema, Rotterdam, 517-523.
17. Kueper, B.H., Haase, C.S. and King, H. 1992. Leakage of DNAPL from waste impoundments constructed in fractured rock or clay: Theory and case History, Canad. Geotech. J., **29**, 234-244.
18. Lenormand, R., Touboul, E. and Zarcone, C. 1988. Numerical models and experiments on immiscible displacements in porous media. J. Fluid Mech., **189**: 165-187.
19. Li, L., Barry, D.A., Hensley, P.J. and Bajracharya, K. 1993. Nonreactive chemical transport in structured soil: The potential for centrifuge modelling. In *Geotechnical Management of Waste and Contamination* (eds R. Fell, T. Phillips and C. Gerrard), Balkema, Rotterdam, 425-431.
20. Li, L., Barry, D.A., Culligan-Hensley, P.J. and Bajracharya, K. 1994a. Mass transfer in soils with local stratification of hydraulic conductivity, Water Resour. Res., **30**(11), 2891-2900.
21. Li, L., Barry, D.A., and Stone, K.J.L. 1994b. Centrifugal modelling of nonsorbing, nonequilibrium solute transport in a locally inhomogeneous soil. Canadian Geotechnical Journal, **31**(4):471-477.
22. Mayer, A.S. and Miller, C.T. 1992. The influence of porous media characteristics and measurement scale on pore-scale distributions of residual nonaqueous phase liquids. J. Contam. Hydrol. **11**:189-213.
23. Mitchell, J.K., 1993. *Fundamentals of Soil Behaviour*, 2nd Ed., Wiley, New York.
24. Morrow, N.R. and Songkran, B. 1981. Effect of viscous and buoyancy forces on nonwetting phase trapping in porous media, In D.O. Shah (ed), *Surface Phenomena in Enhanced Oil Recovery*: Plenum Press: N.Y. 387-411.
25. Naar, J., Wygal, R.J. and Henderson, J.H. 1962. Imbibition relative permeability in unconsolidated porous media, *Soc. Pet. Eng. J.*, 2(13): 225.

26. National Research Council (NRC) 1994. Alternatives for Groundwater Cleanup. National Research Council Committee on Groundwater Cleanup Alternatives, TD426.A48, National Academy Press: Washington.

27. Pahwa, A. 1987. Quick release tests on centrifuge of wall retaining saturated sand backfill. Master of Science Thesis. Department of Civil and Environmental Engineering, Massachusetts Institute of Technology, Cambridge MA.

28. Penn, M. 1997. Remediation of fine grained soils using electrokinetics. Forthcoming PhD. Thesis, University of Cambridge, UK.

29. Penn, M. and Savvidou, C. 1996. Centrifuge modelling of the removal of heavy metal pollutants using electrokinetics. *Proceedings of the Second International Congress on Environmental Geotechnics: IS-Osaka ë96*, Japan, 5-8 November 1996. In Press.

30. Ratnam, S., Culligan-Hensley, P.J. and Germaine, J.T. 1996a. LNAPL entrapment under hydraulic flushing, *Proceedings of the Second International Congress on Environmental Geotechnics: IS-Osaka ë96*, Japan, 5-8 November 1996. In Press.

31. Ratnam, S., Culligan-Hensley, P.J. and Germaine, J.T. 1996b. Modeling the behavior of LNAPLS under hydraulic flushing, *Non-Aqueous Phase Liquids (NAPLs) in the Subsurface Environment: Assessment and Remediation*, *ASCE 1996 National Convention*, November 10-14, 1996. In Press.

32. Schofield, A.N. 1980. Cambridge geotechnical centrifuge operations. Twentieth Rankine Lecture. Geotechnique, **30**(3): 227-268.

33. Smedt, F. De and Wierenga, P.J. 1979. A generalised solution for solute flow in soils with mobile and immobile water. Water Res. Res., **15**(5), 1137-1141.

34. USEPA. 1992. *Evaluation of Ground-Water Extraction Remedies*, Phase II, Vol. 1 - Summ. Rep. Publ. 9355.4-05A. Washington, D.C. EPA Office of Emergency and Remedial Response.