CHEMICAL CHARACTERISATION OF LANDFILL LEACHATE AND ITS POTENTIAL MOBILITY THROUGH THE CAPE FLATS SAND

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ABSTRACT

Researchers have expressed concern about pollution of groundwater at Coastal Park, a large, general waste landfill situated on the False Bay coastline above the Cape Flats Aquifer. The landfill was constructed without a liner, but with an average 2 m separation of calcareous sand providing a "buffer" zone between the waste pile and the water table. Water balance studies and application of a model, FLOW, have predicted that leachate will be generated seasonally. This study was initiated as a result of uncertainties about hydrological and geochemical aspects, such as the hydraulic conductivity of the soil in the buffer zone and the degree of leachate attenuation occurring in this zone.

The Coastal Park soil was classified as an aeolian, calcareous, medium quartzitic sand with negligible organic carbon content. Extreme clay-depletion would render the soil almost incapable of leachate attenuation, although calcite and aragonite, found by X-ray diffractometry, would impart a significant pH buffering capacity to the soil.

The solid phase of a locally-derived landfill leachate (sampled from Vissershok landfill, about 35 km NW of Cape Town) was found to contain amorphous sulfides of iron and heavy metals, and green rusts which are mixtures of Fe^{2+} - Fe^{3+} hydroxides, in addition to organic matter. The solid phase was isolated by centrifugation, freeze-dried, and analyzed by XRF and XRD. Distribution coefficients of heavy metals in the leachate (at pH 7.7) demonstrated the high affinity of heavy metals, such as Cu, Zn, Cr, Ni and Pb, for the solid phase. The leachate solid phase consists of amorphous solids, with high Ca and Cl concentrations in the liquid phase leading to halite and calcite formation upon evaporation of the liquid phase.

According to locally specified requirements by Department of Water Affairs and Forestry, a landfill liner material must have a hydraulic conductivity (K) not higher than 1×10^{-7} $cm.s^{-1}$. Air dried samples of Coastal Park soil were treated with various amendments to test their efficacy as landfill liners. An 8 % kaolinite plus 4 % gypsum treatment was the most effective, maintaining a minimum K of $10^{-4.5}$ $cm.s^{-1}$, which, however, is still higher than the local requirement. Amendment with 8 % Na-bentonite initially achieved a minimum K of $10^{-7.8}$ $cm.s^{-1}$, but the high electrical conductivity (EC) of the leachate (26.8 $mS.cm^{-1}$) caused shrinking and severe side-wall seepage, which rapidly enhanced hydraulic

conductivity, reaching a maximum K of about $10^{-4.7}$ cm.s^{-1}. Both treatments of the sand do show promise as possible liners, although the use of higher percentage concentrations of clay should be investigated further.

LEACHW (the water regime submodel of LEACHM) was used to predict leachate discharge from the Coastal Park landfill, assuming a hypothetical capping system of 1 or 2 m soil depth with 0, 50, 70, or 90 % vegetation cover (*Acacia cyclops*), and based on the assumption that drainage from this layer into the waste pile contributes directly to leachate generation. The model predicted that under average rainfall conditions the landfill, with a 2 m soil depth and 0 % vegetation cover, would not generate leachate. However, under the wettest conditions not even a 90% vegetation cover and 2 m soil cover would be sufficient to prevent the landfill from generating leachate, suggesting that, under such conditions, a more effective leachate management strategy, such as leachate collection sumps, should be implemented. This exercise demonstrated the use of LEACHM as an alternative means of predicting leachate discharge from landfill sites.

TABLE OF CONTENTS

Chapter 2

Soil characterisation and geochemical analysis of landfill leachate

Chapter 3

Factors affecting hydraulic conductivity of Coastal Park soil

Chapter 4

Modelling the water balance and leachate generation at the Coastal Park landfill using LEACHM

TABLE OF FIGURES

INTRODUCTION

The positioning of the Coastal Park landfill on the highly permeable calcareous sands above the Cape Flats Aquifer, the exclusion of an adequate liner, and the likelihood of leachate generation, has prompted a geochemical and hydrological investigation into the potential for movement of leachate through the Cape Flats soil. Water balance studies carried out at the inception of the Coastal Park project in 1985, warned that the landfill would produce leachate seasonally (Ball and Blight, 1995). However, on account of the seaward groundwater flow direction and the current belief then that a 2 m "buffer" zone of calcareous sands below the landfill and the sea would attenuate any pollution by leachate, the landfill was established (Ball and Blight, 1995). Monitoring leachate generation would facilitate remedial action in the event of pollution becoming unacceptable. Questions regarding the extent of attenuation of the leachate by the calcareous sand and *in situ* hydraulic conductivities controlling the migration of the pollution plume were raised (Blight, Ball and Vorster, 1994; pers. comm. Ball, 1995). This project was not aimed at finding a solution to a practical problem, but rather, its objective was to provide information of geochemical and hydrological significance which may be used in the design and construction of future extensions, for example, a liner or capping system, at Coastal Park and elsewhere on the Cape Flats.

Soils which are not suitable for leachate containment in their natural state can be manipulated by inducing dispersion and subsequent flocculation of added clay minerals (Van Olphen, 1977; Smith and Fey, 1993; McBride, 1994; Nowicki and Fey, 1994). One of the aims of the project was to investigate, on a laboratory scale, a possible means of creating a low-permeability layer of soil at Coastal Park, and to assess the behaviour of leachate in this layer. A review of the literature on principles of hydraulic conductivity and influences of clay systems and colloidal chemistry on the flow of liquids through soils has been undertaken. Exchangeable cations, solution ionic strength, mineralogy, pH and specifically adsorbed anions and organic matter are some of the factors which can increase or decrease the hydraulic conductivity of a soil (Clem, 1985; Brown, 1986; Frenkel, Levy and Fey, 1992a, 1992b). This investigation has shown, using leaching columns, how swelling clays differ from non-swelling clays in their effect on reducing hydraulic conductivity of a sandy soil. Research by Nowicki and Fey (1994) and Smith

and Fey (1993), into chemical manipulation of soils as landfill liners, formed the basis for the approach to the column work. Kaolinite and bentonite clay amendments and leaching solutions, including landfill leachate, were tested for their effect on the hydraulic conductivity of the Cape Flats soil.

Standard analytical techniques were employed for the analysis of soil sampled from Coastal Park and landfill leachate from Vissershok landfill. An investigation into the qualitative properties of the leachate, such as its colour, has been conducted and the information interpreted in the context of its chemical composition and oxidative state. A new approach employed in this study, in which the leachate solid phase was isolated by centrifugation, which differed from methods used by Gounaris *et al.* (1993) who made use of ultra- and microfiltration for isolation of leachate solid phase size fractions. Also, freeze-drying has been used in preparation of the solid phase for X-ray diffraction analysis, a method not mentioned in any of the literature reviewed.

The final aspect of the project involved the use of LEACHW, the water regime submodel of LEACHM, to estimate the water balance of a hypothetical capping system at Coastal Park. Blight (1995) used real-time data to empirically estimate the water balance of the *landfill.* A different approach has been used here in which LEACHM estimates the water balance of the *capping system,* utilising refined equations (eg. Richards equation) to calculate transient flow of water under saturated and partially saturated conditions. The results are interpreted on the basis of the assumption that a positive water balance of the capping system would contribute directly to leachate generation at the base of the landfill. The water balance is simulated using real-time daily climatic data and varying degrees of vegetation cover and capping soil depth. The results are interpreted on the basis of the assumption that a positive water balance of the capping system would contribute directly to leachate generation at the base of the landfill. LEACHM can be considered a sophisticated and alternative approach to water balance studies at Coastal Park.

The overall objectives of the project are therefore summarised as follows:

1) To review the literature pertinent to hydraulic conductivity of soils, landfill liners and associated materials, and the physicochemical factors influencing it.

2) Characterisation of the Cape Flats sandy soil, geochemical analysis of a locally derived landfill leachate, and the use of this information in explaining the hydraulic behaviour of the soil in laboratory permeameter tests.

3) Establish and interpret the effects of a locally derived kaolinite and a commercial bentonite clay as soil additives, together with various leaching solutions, on the hydraulic conductivity of the sandy soil.

4) Model the water balance of a hypothetical capping system which would be required in the event that Coastal Park receives a permit of closure from Department of Water Affairs and Forestry (*Minimum Requirements,* 1994). The LEACHM model will be used to simulate the water balance on the basis of a broad set of assumptions in which a positive water balance of a capping system would contribute directly to leachate generation.

Chapter 1

Chemical and mineralogical factors affecting hydraulic conductivity of soil materials: A literature review

1.1 Introduction

The rate at which water moves through soil is not only a function of its viscosity and density, but also the physicochemical interactions taking place between the soil particles and the pore fluid. This review of available literature focuses on these physicochemical interactions, the most important of which are the swelling and dispersive properties imparted to soils by the expanding 2:1 clay minerals.

Aqueous-solution chemistry affects interaction forces between clay particles, and thereby influences both flocculation and dispersion. It is important to quantify how chemistry affects both of these processes to help control decline in soil permeability and colloid-facilitated contaminant transport. The study of these processes finds application in waste disposal site subsurface liners. In their performance, clay liners function by two mechanisms: (1) impedance of flow of the pollutant carrier (usually water) into the subsoil, or (2) attenuation of suspended or dissolved pollutants so that the exiting leachate contains contaminants within acceptable groundwater limits, or both of these mechanisms in combination (Eklund, 1985).

The adsorptive or attenuation capability of the liner depends on the chemical composition and mass of the liner. Some soils do not provide an impermeable boundary but still possess significant attenuating properties and an adsorbing capacity for different contaminants. These properties can sometimes be enhanced by the use of soil additives. A good soil liner will contain a minimum of 25 to 28% clay-size particles by weight (Eklund, 1985). The clay fraction (<0.002 mm) has a relatively large surface area, and the mechanical behaviour of a clayey soil is dependent upon the physicochemical interactions between the soil particle surfaces. The permeability of the soil is dependent upon both the soil and leachate properties. A good liner will usually possess a hydraulic

conductivity coefficient (K) of 1 x 10^{-7} $cm.s^{-1}$ or less when it is tested with a simulated (or actual) waste fluid (Gipson, 1985).

For nonhazardous wastes a desirable disposal practice is to utilize the natural attenuation processes of the soil media beneath the disposal site. These attenuation processes include precipitation brought about by changes in the aqueous environment, adsorption onto soil surfaces, filtration of contaminants by tightly packed soils, and dispersion and diffusion in the vadose zone and in the groundwater. Some species, for example chloride, often migrate through the soil at the same velocity as the transport fluid. No attenuation occurs in this case.

In this review a discussion of Darcy's Law and associated concepts precedes the main focus on the physicochemical properties, including swelling, dispersion, mineralogy, pH and specifically adsorbed anions and organic substances, which have an influence on soil hydraulic conductivity. Even though unsaturated hydraulic conductivity is equally important, if not more so, than saturated flow conditions in the field, the latter is concentrated on in this review. The reason for this approach is that the motivation for the review is founded on work relating to waste liquid containment below landfill disposal sites, in which saturated conditions are assumed. Also, a very important factor affecting flow of water through soils is biological activity. Time and volume constraints have prohibited the inclusion of a formal section on biological activity in this review. However, it is acknowledged that such activity may influence hydraulic behaviour of soils, particularly in the case of landfill sites where microbial populations can have an impact on pore-space geometry, as well as temperature and chemical effects which can influence fluid densities and contaminant mobility.

1.2 Water movement in soils - Darcy's Law and hydraulic conductivity

1.2.1 General principles

Hydrodynamic processes include those phenomena that result from the physical movement of fluids in the subsurface. The soil properties that determine the behaviour of soil water flow systems are the hydraulic conductivity and water retention

characteristics. The *hydraulic conductivity* of the soil is a measure of its ability to transmit water; the *water retention characteristic* is an expression of its ability to store water. These properties determine the response of a soil water system to imposed boundary conditions. In some cases, either the hydraulic conductivity or the water *diffusivity*, which is the ratio of hydraulic conductivity to the differential water capacity, may be used to analyze the behaviour of a soil water system. These properties are often called the *hydraulic properties of the soil* (Klute and Dirksen, 1986).

The conductivity of the soil depends on the geometry of the pores and the properties of the fluid in them. The two fluid properties that directly affect the hydraulic conductivity are viscosity and density. The texture and structure of the soil are the principal determinants of the geometry of the water in the soil pores. In soils with appreciable clay content, the composition of the soil solution can significantly affect the hydraulic conductivity because of interactions between the soil solution and the soil matrix (Klute and Dirksen, 1986).

1.2.2 Darcy's Law

Darcy's Law is the basic equation governing fluid flow through a *saturated* porous medium, and can be represented as:

$$V = -KS \qquad (1)$$

where V = specific discharge (Darcy velocity) of fluid [L/T]
K = hydraulic conductivity tensor [L/T]
S = hydraulic gradient (dimensionless)
L,T = dimensions of space and time

Certain points need to be highlighted regarding Darcy's Law and hydrodynamic processes governing the fate and transport of contaminated groundwater. Firstly, the Darcy velocity, V, is not the actual velocity of the fluid in the porous medium. V is an average or discharge velocity based on volumetric flow. To calculate the actual pore water velocity of the fluid Equation 2 can be applied:

$$V_s = \frac{V}{\eta} = -\frac{KS}{\eta} \qquad (2)$$

where Vs = pore water velocity (L/T)

η = porosity of the porous medium (dimensionless), required in most transport and fate analyses.

Secondly, K in Equation 1 is a function of both the saturated formation and the flowing fluid (water). K is a tensor quantity and thus possesses magnitude in each of the orthogonal (x,y,z) directions.

The third point to note regarding Equation 1, is that, in the strictest sense, the hydraulic gradient (S) is applicable only on streamlines. The hydraulic gradient represents the rate of change in total head (elevation plus pressure) with respect to a specified reference point.

A more general form of Darcy's Law can be written with differentials:

$$V_l = -K_l\left(\frac{\delta h}{\delta l}\right) \qquad (3)$$

where Vl = Darcy velocity in the "l" direction

K = hydraulic conductivity in the "l" direction

$\delta h/\delta l$ = hydraulic gradient in the "l" direction

The negative sign preceding K in Darcy's Law is needed because derivatives are taken in the direction of flow, i.e., lower total head minus higher total head. The negative sign ensures a positive velocity from a high total head to a low total head.

Klute and Dirksen (1986) explain that the driving force (H) is expressed as the negative gradient of the hydraulic head composed of the gravitational head, z, and the pressure head, h; i.e.,

$$H = h + z \tag{4}$$

The pressure head has two components, a matric (or suction) pressure head, hm, and a pneumatic pressure head, ha. The latter represents the (possible) difference between the soil gas-phase pressure and the atmospheric pressure.

Note that Darcy's Law governs flow of a fluid at a point in a saturated porous medium. There are other, more complex equations which can be used to represent the groundwater flow behaviour of an aquifer acting as a whole unit. Groundwater velocity tends to dominate the hydrodynamic subsurface contaminant transport processes and can influence the degree to which the other processes and/or reactions can occur (Knox *et al.*, 1993).

The *intrinsic permeability* is related to the hydraulic conductivity through Equation 5 and is a physical property specific to the porous medium:

$$k = K\left(\frac{\mu}{\gamma}\right) = K\left(\frac{\mu}{\rho g}\right) \tag{5}$$

where k = intrinsic or physical permeability of the porous medium (L2)

μ = dynamic viscosity of the fluid (M/LT)

γ = unit weight of fluid ($M/L^2/T^2$)

ρ = mass density of the fluid (M/L^3)

g = gravitational acceleration (L/T^2)

In the case of immiscible fluids or highly contaminated water, the density and viscosity of the fluids will be different from that of water and can, according to Equation 5, significantly affect transport of the fluid. In these situations, the coefficient of permeability is required for the assessment of the fluid conductive properties of the

porous medium. The dimensions of the intrinsic permeability are length squared. The above equation assumes that the properties of the fluid are not affected by the nature of the solid matrix, and that the intrinsic permeability is a function of only the pore-space geometry. According to these assumptions, the same intrinsic permeability will be obtained for different fluids. This ideal is seldom realised in soils, and changes in the fluid will generally affect the matrix. Changes in the concentration and kinds of cationic species in the soil solution cause large changes in the hydraulic conductivity of soils containing significant amounts of clay, especially swelling clays (Klute and Dirksen, 1986).

1.2.3 Saturated versus unsaturated flow

Unsaturated flow conditions are in general complicated and difficult to describe quantitatively, since they often entail changes in the state and content of soil water during flow. Such changes involve complex relations among the variables soil wetness, suction and conductivity, whose interrelations may further be complicated by hysteresis. In recent decades, unsaturated flow has become one of the most important and active topics of research in soil physics, which has resulted in significant theoretical and practical advances (Hillel 1982).

Soil water flow is caused by a driving force resulting from a potential gradient and flow takes place in the direction of decreasing potential. The rate of flow (flux) is proportional to the potential gradient and is affected by the geometric properties of the pore channels through which the flow takes place. These principles apply to both saturated and unsaturated flow (Hillel, 1982). The moving force in a saturated soil is the gradient of a positive pressure potential. On the other hand, water in an unsaturated soil is subject to a subatmospheric pressure, or suction, which is equivalent to a negative pressure potential, the gradient of which constitutes a moving force. Matric suction is due to the affinity of water molecules for soil particle surfaces and to capillary forces. Water will tend to move from a zone where the capillary menisci are less curved to where they are more highly curved, or from where the hydration envelopes surrounding the soil particles are thicker to where they are thinner. The moving force is greatest at the wetting front zone, where water is advancing into an originally dry soil, the region where the suction gradient is at its greatest.

Unsaturated flow can be illustrated schematically in Figure 1.1 with the associated interrelations in unsaturated flow in Figure 1.2.

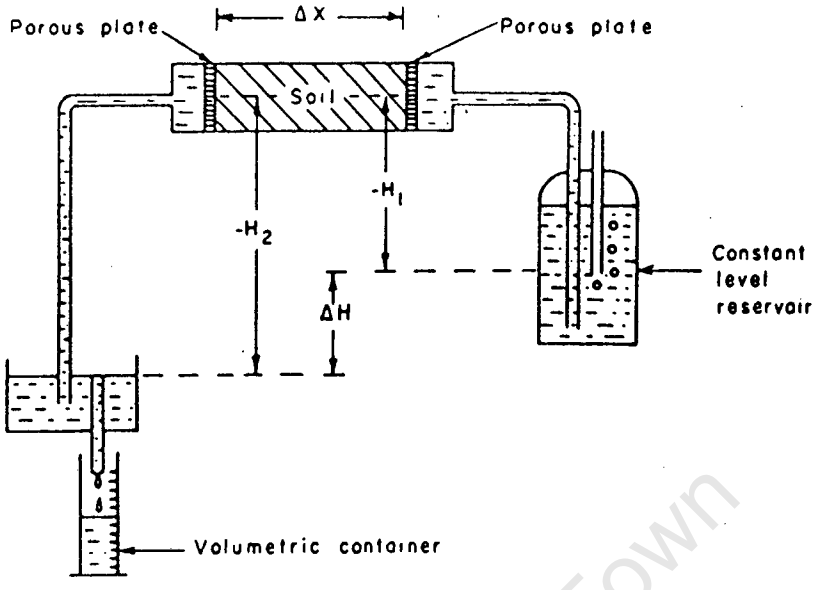

Figure 1.1 A model illustrating unsaturated flow (under a suction gradient) in a horizontal column (Hillel, 1982).

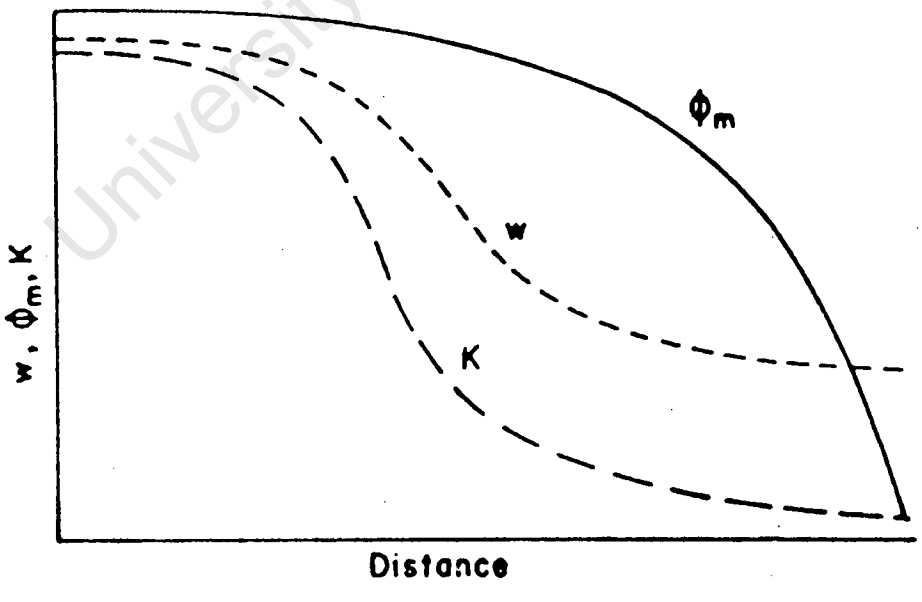

Figure 1.2 The variation of wetness *w*, matric potential ϕ_m, and conductivity K along a hypothetical column of unsaturated soil conducting a steady flow of water (Hillel, 1982).

The hydraulic conductivity is perhaps the most important difference between saturated and unsaturated flow. All the pores are filled and conducting under saturated conditions, so that continuity and conductivity are maximal. There is a corresponding decrease in the cross-sectional area of the soil's conducting portion as the soil desaturates and some of the pores become air filled. In addition, it is the largest pores which are the most conductive, but which are also the first to empty. A consequence of desaturation is an increase in tortuosity (Figure 1.3) because of the circumvention of empty pores.

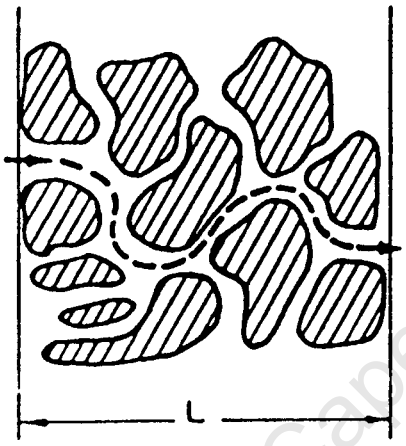

Figure 1.3 Flow path tortuosity in the soil (Hillel, 1982).

At saturation, the most conductive soils are those in which large and continuous pores constitute most of the overall volume, while the least conductive are the soils in which the pore volume consists of numerous micropores. Thus a saturated sandy soil conducts water more rapidly and with greater ease than a clayey soil. However, the opposite can be true for unsaturated conditions. In a soil with large pores, these pores empty quickly and become nonconducting as suction develops, thus steeply decreasing the initial high conductivity. On the other hand, in a soil with small pores, many of the pores retain and conduct water even at appreciable suction, so that hydraulic conductivity does not decrease as steeply and may actually be greater than that of a soil with large pores and subjected to the same degree of suction.

In most instances the soil in the field is in an unsaturated state, and it often happens that flow is more appreciable and persists longer in clayey than in sandy soils. For this reason a layer of sand in a fine-textured profile, far from enhancing flow, may actually impede unsaturated water movement. Only when water accumulates above the sand and suction decreases sufficiently for water to enter the large pores of the sand, can this impedance be overcome (Hillel, 1982)

1.2.4 Methods of measuring hydraulic conductivity

Several methods are available for the measurement of hydraulic conductivity and diffusivity. The choice of method is a function of (i) the availability of equipment, (ii) the nature of the soil, (iii) the kind of samples available, (iv) the skills and knowledge of the experimenter, (v) the soil water suction range to be covered (if unsaturated conditions prevail), and (vi) the purpose for which the measurements are being made.
Fixed-wall and flexible-wall cells are two general types of permeameters which can be used to measure the hydraulic conductivity of relatively impermeable, fine-grained soils (Figures 1.4 and 1.5 from Daniel *et al.*, 1985).

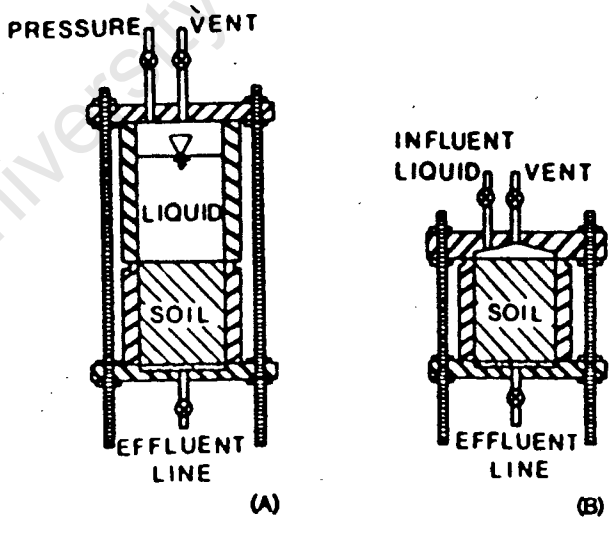

Figure 1.4 Compaction-mold permeameters with (a) a reservoir of permeant liquid contained with a collar located directly above the soil specimen and (b) a separate reservoir of permeant liquid (Daniel *et al.*, 1985).

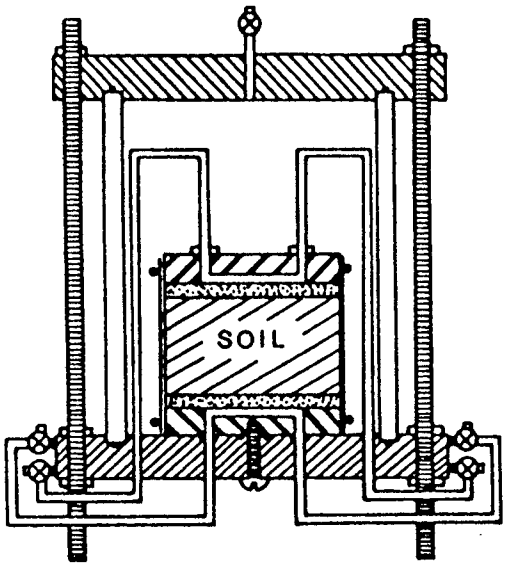

Figure 1.5 Flexible-wall permeameter (Daniel *et al.*, 1985).

In the past there has been some controversy over which cell is best suited for permeability measurements of such soils. Daniel *et al.* (1985) showed that the type of permeameter used has little effect on laboratory compacted clay permeated with water, but can have a major effect on measurements for clay permeated with concentrated organic chemicals.

Anderson *et al.* (1985), on the basis of double-ring permeameter tests, showed that permeability values determined using 0.05 M $CaSO_4$ solution as the permeant liquid may be misleading. They concluded that large permeability increases were obtained when clay soil-bentonite mixtures were permeated by organic liquids. This occurred with both a nonpolar liquid (xylene) and a polar liquid (methanol). Permeability increases occurred in both the inner and outer chambers of the double-ring permeameter. Anderson *et al.* (1985) also showed that it was possible to differentiate flow near the sidewalls from flow through the central portion of the specimens, thus giving a more accurate value for the permeability than could otherwise be obtained.

Daniel *et al.* (1985) point out the relative advantages and disadvantages of each type of permeameter. With rigid-wall permeameters, the major advantages are low cost, simplicity, applicability to testing compacted soils, compatibility with a wide range of permeant liquids, and a lack of need to apply high confining pressure. The major disadvantages are incomplete control over stresses, inability to measure deformations in

most fixed-wall cells, difficulty in trimming soil samples into the containing rings, and potential for side wall leakage. Use of double-ring compaction-mold permeameters shows promise for minimising uncertainties associated with sidewall leakage.

Flexible-wall permeameters have the advantage of minimising sidewall leakage, permitting control over vertical and horizontal stresses, enabling one to measure vertical and volumetric deformations, permitting convenient back pressuring of the soil, enabling one to measure saturation, and providing the possibility of testing specimens with a range of diameters in the same cell. The disadvantages are higher cost, compatibility problems between the flexible membrane and some permeant liquids, and, perhaps most importantly, a need to apply significant confining pressures at high hydraulic gradient in order to maintain contact between the membrane and the soil (Daniel *et al.*, 1985).

Daniel *et al.* (1985) suggested that fixed-wall cells are perhaps best suited to testing laboratory-compacted clays that will be subjected to little or no effective burden pressure in the field. These cells are better suited to testing undisturbed samples of soil (to minimise boundary leakages) and testing soils that will be subjected to significant effective stress.

The method in which the soil specimen is prepared for permeability measurements is important. Test results by Edil and Erickson (1985) indicated that bentonite-sand specimens continue to hydrate during permeation. Unless wetter specimens are used, this continuing hydration interferes with the inflow-outflow balance, depriving the tester of an important check for leaks. Differential hydration throughout the specimen results in different soil structures and zones of flow. For instance, most of the flow may take place in an annular area surrounding a less hydrated core in the centre. Since the total cross-sectional area is used in computing the coefficient of permeability, the values may be underestimated. Back-pressure, often used to enhance saturation during testing, appears to have a detrimental effect when applied to the rigid-wall permeameters by increasing the potential for channels and sidewall flow. Testing results are also affected by hydraulic gradient in different ways depending on the type of permeameter. While gradients as high as 360 did not induce piping in the gap-graded liner material tested, the application of very high gradients to accelerate testing is not desired for a number of

other effects observed (Edil and Erickson, 1985). The permeabilities measured in flexible-wall permeameters decreased as much as 58% when hydraulic gradients were increased, and was thought to be caused by physical changes such as differential consolidation of the specimen. Another possible reason for the reduced permeability was that the permeant was probably flowing through a highly hydrated area near the specimen edges resulting in a nonuniform cross-sectional area of flow.

Lentz *et al.* (1985) use the following falling head equation to measure the permeability:

$$K = \left(\frac{aL}{A(t_i - t_f)}\right) \ln\left(\frac{h_i}{h_f}\right) \qquad (6)$$

where k = permeability, $cm.s^{-1}$,

a = cross-sectional area of the standpipe, cm^2,

L = specimen thickness, cm,

A = cross-sectional area of the specimen, cm^2,

$t_i - t_f$ = elapsed time, s,

h_i = initial head difference, cm,

h_f = final head difference, cm.

1.3 Factors affecting hydraulic conductivity

1.3.1 Physical properties - swelling and dispersion

In the literature, smectite, illite and kaolinite are the main minerals which have come under investigation. Of these, the 2:1 smectitic minerals are the most reactive in the environment and are commonly the most responsive to changes in the soil solution chemistry. Smectite is able to absorb an amount of water equivalent to several times its own weight. The small size, large surface area and negatively charged layer structure of smectite allow for its high water absorption capacity relative to other clay minerals (Borchardt, 1989).

At high water contents and large interlayer distances, the double-layer theory provides the best available explanation for describing the influence of solution environment on swelling. Increases in the concentrations of exchangeable cations in the solutions bathing smectites tends to collapse the layers, while decreases tend to expand them (Borchardt, 1989).

Borchardt (1989) explains that there are two primary viewpoints from which the remarkable swelling properties of smectites have been explained. The first, in Sposito (1973) and Sposito *et al.* (1983), emphasises the hydration and mobility of the cations that balance the negative charge of the layer silicate. With reference to Low (1981), the second viewpoint emphasizes the interaction of H_2O molecules with the silicate layer. However, Borchardt (1989) states that both viewpoints fit the common observation that smectitic soils shrink when H_2O is lost from the bulk solution or when salts are added to it.

Norrish (1954), according to Borchardt (1989), showed that Ca-saturated smectite swells from 1.0nm to a maximum of 2.0nm. On the other hand, Na- and Li-smectite theoretically swell to infinity. The liquid limit (LL) is the moisture content necessary for a soil sample to flow along a 1:1 slope (Borchardt, 1989). At the LL Na-montmorillonite may have H_2O films between 10.0nm and 20.0nm thick. Borchardt (1989) emphasizes here that smectites in soils do not expand nearly as much as this for two reasons: (i) they are often interstratified with other nonexpansive clay minerals such as micas, chlorites, kaolinites and chloritic interlayers; and (ii) they are seldom monoionic and usually saturated with Ca, Mg and K rather than Na or Li. Borchardt (1989) goes into some detail about how the LL and plasticity index of the soil relate to the quantity of smectite present in the soil and how these measurements are strongly influenced by the chemical environment and cation saturation of the smectite.

Borchardt (1989) explains how the expansive properties of smectitic soils may be ameliorated by the treatment with lime or other chemicals. Lime produces a dramatic decrease in the plasticity index (plasticity index = liquid limit minus plastic limit): first, by increasing the plastic limit, and second, by producing hydrated Ca-Al silicate cementing agents. The cementing agents are produced by the usual pozzolanic reactions

that occur during the first week after lime is added to materials containing Al, Si, and H_2O.

In their study on the effect of electrolyte concentration on soil permeability, Quirk and Schofield (1955) suggested that three processes were operative:
(i) *Swelling* results in complete or partial blocking of the larger conducting pores. Flow of water through a pore is proportional to the fourth power of the pore radius, therefore appreciable decreases in permeability may result even when effects from swelling are relatively small.
(ii) *Failure* of the soil aggregates can occur because of stress resulting from unequal swelling throughout the soil mass. In this regard, organic matter is capable of acting to prevent failure.
(iii) *Deflocculation*. During processes of swelling the particles can separate past the critical distance which allows the interparticle repulsive forces to dominate the van der Waals attractive forces which leads to dispersion (Tan, 1992; van Olphen, 1977; Wild, 1994). Deflocculation and flocculation are two opposing colloidal processes involving clay particles which may influence the physical or chemical properties such as hydraulic conductivity or contaminant transport adsorbed to migrating particles.

Experimental evidence by Frenkel *et al.* (1978) showed that clogging of pores in montmorillonite, vermiculite and kaolinitic soils was caused by dispersed clay particles at varying exchangeable sodium percentage (ESP) values and salt concentrations (in the range of SAR and electrolyte concentration of 0-10 and 10-30 meq/l, respectively). Dispersion was considered the major cause of hydraulic conductivity reduction. A change of hydraulic conductivity due to swelling is essentially a reversible process whereas changes due to dispersion and particle movement are irreversible.

1.3.1.1 Diffuse double layer theory

Tan (1992) gives a fairly simplistic explanation of the electric double layer concept, upon which the ensuing discussion is based.
Generally clays carry a negative charge which is ordinarily balanced by cations adsorbed on their surfaces. In suspension, cations diffuse away from the clay surface in response

to the concentration gradient between the clay surface and the bulk of the solution. However, a large portion of these cations, especially those in the immediate vicinity of the clay particle surface, cannot move far away from the surface because of their strong electrostatic attraction for the negative surface. The cations aggregate at the interface, in response to this electrostatic attraction, thereby forming a diffuse electric double layer, which can vary in thickness from 50 to 300 Å. When two such particles approach one another, they experience mutual repulsion because of the positively charged outer region of the double layers belonging to each surface. In this situation the particles are considered to be dispersed and a stable suspension is said to exist. As the two particles approach each other further, their diffuse counterion atmospheres interfere with one another which leads to a rearrangement of the ion distribution in the double layers of both particles. Work, in the form of the *repulsive energy* or the *repulsive potential*, V_r, must be done on the system to bring about these changes at a given distance. Figure 1.6 illustrates the exponential relationship between V_r and the distance from the surface of a particle. At a closer interparticle distance and opposing V_r is the attractive or *van der Waals* force, V_a, which decays rapidly with distance from the surface (Figure 1.6).

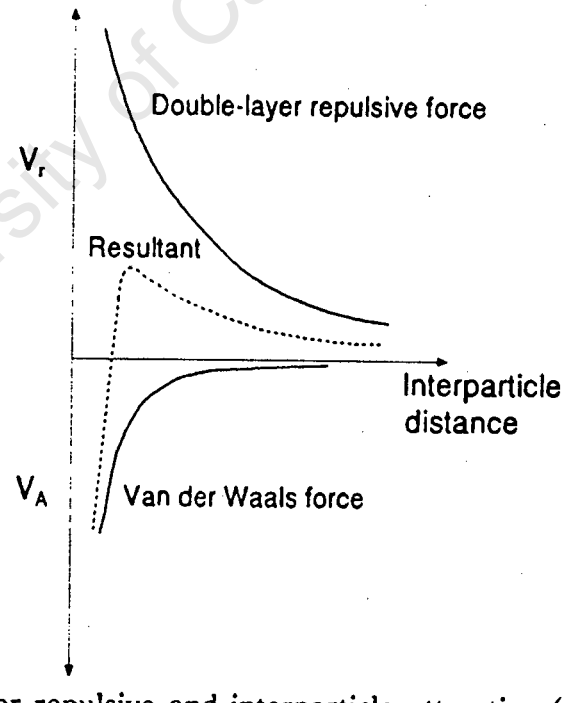

Figure 1.6 Double-layer repulsive and interparticle attractive (van der Waals) forces as a function of interparticle distance (Tan, 1992).

These potentials are additive and give rise to the resultant force acting between the particles. When the interparticle distance decreases to about 20 Å or less, V_a dominates and the clay particles undergo flocculation. V_r dominates at interparticle distances greater than 20 Å, leading to dispersion and a stable suspension.

Wild (1994) adds that repulsion dominates at low electrolyte concentrations when clay particles are shielded by relatively thick double layers, decreasing the possibility of mutual approach. At high electrolyte concentrations compression of the double layer allows mutual approach to take place and $V_a > V_r$, resulting in coagulation or flocculation of the clay particles. The diffuse double layer becomes compressed in solutions of high electrolyte concentration, mainly because diffusion away from the surface is less.

Properties of the cation also have an effect on the dispersive properties of a clay: (i) the higher the charge the greater is its attraction to the particle surface, and (ii) the greater the hydration, the weaker is its attraction to the particle surface. For example, Na is weakly held because of its monovalent positive charge and because of its high degree of hydration. On the other hand, Ca is strongly held because of its divalent positive charge, lower degree of hydration and subsequently more compact diffuse double layer.

Consider the distribution of Na^+ ions in a 2:1 sodium-saturated clay suspension. As the particles in the suspension approach one another, their diffuse double layers overlap leading to an increase in Na^+ ion concentration in this region. A tendency for water to migrate into this region in an attempt to restore the original distribution of ions develops, and the clay particles effectively repel one another in the process (Wild, 1994). When the diffuse layer is highly compressed, e.g. high electrolyte concentration or by replacing the Na^+ with Ca^{2+} on the exchange complex, the clay particles can approach close enough to allow them to aggregate or flocculate.

Variations in absolute values of hydraulic conductivity indicate differences in the structure of the compacted soil (Acar *et al.*, 1985a) . Structural changes are initiated by changes in the surfaces forces of interaction due to variations in the attractive and repulsive forces between clay minerals. Repulsive forces are primarily attributed to the interaction between diffuse double layers. The system variables that control these forces are defined

in the development of the Gouy-Chapman theory of diffuse double-layer around the clay minerals (van Olphen, 1977). An approximate quantitative indication of the thickness of the double-layer can be expressed as :

$$H = f\left[\left(\frac{DT}{h_0^2 v^2}\right)^n\right] \qquad (7)$$

where H = relative thickness of the double layer,
D = dielectric constant of the medium,
T = temperature,
h_0 = electrolyte concentration,
v = cation valence, and
n = a constant (approx. = 1/2).

Changes in the thickness of the diffused double layer are directly related to the forces of repulsion between the clay particles. The principal contribution to attractive forces arise from van der Waals forces, which can be either ion-dipole or dipole-dipole interactions. Additional attractive forces arise from Coulombic attractions between negative surfaces and positive edges, cation linkages, and hydrogen bonding (Lambe, 1982; cited in Acar *et al.*, 1985a). Net forces of interaction are functions of the static dielectric constant of the pore fluid. It is well recognised that changes in the dielectric constant of the medium would re-establish the net forces of interaction resulting in variations in the structure of the soil.

Deflocculation arises from changes in soil solution chemistry that reduce the interparticle attractive forces within the flocs, thereby causing the particles to break apart either spontaneously or with mechanical energy inputs, such as raindrop impact or possibly fluid flow. A consequence of deflocculation is the production of a suspension in the soil solution.

Results of a model describing suspended-particle entrapment in porous media (Rege and Fogler, 1987) indicated that hydraulic conductivity decline induced by particle trapping

in pore throats depend on the relation between the particle-size and the pore-size distribution (Hesterberg and Page, 1993). Flocculation coarsens suspended-particle size distribution and decreases the number of particles at a rate depending on solution and surface chemistry (Overbeek, 1952, and van Olphen, 1977; in Hesterberg and Page, 1993), while deflocculation increases the fineness and number of particles. In this light. the rate at which suspensions flocculate (suspension stability) and their floc integrity (resistance to deflocculation) could influence particle trapping in the pore network.

1.3.2 Exchangeable cations

There is concern over the effects of sodium when it occurs in excess of 10 to 20% of the total cation exchange capacity. 10% is the upper limit for fine textured swelling clays. 30% is the upper limit for more sandy soils with swelling clays (Tisdale *et al.*, 1984).

ESP (exchangeable sodium percentage) represents the exchangeable sodium content of a soil relative to the sum of all exchangeable cations:

$$ESP = \left[\frac{Exchangeable\ Na^{+}\ ions}{\Sigma\ Exchangeablecations}\right] \times 100\,\%$$

Saline soils ordinarily have a pH of less than 8.5 and an ESP = 15% above which dispersion of the clay particles can occur. Because of the presence of excess salts and the low amounts of Na^+ in the exchange position, these soils are usually in a flocculated state, and their permeability is considered to be equal to or higher than saline-alkali (ESP > 15%) and sodic soils (ESP >> 15%) (Tan, 1992). Clay swelling is not greatly affected by low ESP values (< 10-15) but at ESP > 15 clay swelling becomes marked (Frenkel *et al.*, 1992; McNeal *et al.*, 1968; Oster *et al.*, 1980; Tisdale *et al.*, 1984).

Selectivity of cation adsorption is greater for divalent than for monovalent cations, and for larger cations than for smaller ones because of the greater degree of hydration of the smaller cations. The typical affinity series can be represented as follows:

$$Al^{3+} > Ba^{2+} > Sr^{2+} > Ca^{2+} > Mg^{2+} = Cs^{+} > K^{+} = NH_4^{+} > Na^{+}$$

Cations held on the surface of soil minerals and within the crystal framework can undergo exchange reactions reversibly in salt solutions and acids. The cation exchange capacity (CEC) is defined as the sum of the exchangeable cations of the soil. A high CEC generally denotes a high clay content and potentially a high capacity to attenuate contaminants. Soils vary in CEC from < 1.0 to > 100meq/100g. Exchangeable bases are defined as the alkali and alkaline earth metals (principally calcium, magnesium, potassium and sodium) that are attached to the clay and organic constituents of the soil. These ions can be exchanged with each other and with other positively charged ions in the soil solution.

Quirk and Schofield (1955) found that with decreasing electrolyte concentration and with increasing exchangeable sodium percentage (ESP), the hydraulic conductivity decreased. There studies were based on the threshold concentration which is equivalent to the concentration of salt which can cause 10-15% (arbitrarily chosen as a reference level) reduction in permeability.

Quirk and Schofield (1955) brought soil pads (40% illite, 40% kaolinite and 20% vermiculite) into equilibrium with the chlorides of Na, K, Mg and Ca cations respectively. This ensured initial cation saturation of the soil, after which progressively more dilute solutions were allowed to permeate each soil. In the case of the Na-saturated soil the major decreases in the permeability took place in the first 2 hours after the solutions were changed. This decrease in permeability was attributed to swelling and deflocculation. Swelling was expected to precede deflocculation, which was observed at 2.5×10^{-2} M NaCl.

Progressively more dilute KCl solutions caused decreases in permeability. The trend parallelled that of the Na-saturated soils, except the decreases occurred at lower concentrations, e.g. 1×10^{-1} M NaCl curve similar to 2×10^{-2} M KCl curve.

Using $MgCl_2$ solutions, decreases in permeability were evident at concentrations below 3.16×10^{-3} M $MgCl_2$. Quirk and Schofield (1955) noted that the threshold concentrations of mixed-ion systems occur when the concentration of divalent cations is reduced to the concentration of the homoionic divalent system. Using four series of Na-Ca solutions,

the threshold concentration of each series occurred when the Na ion concentration in the mixed solution had been reduced to 2.5×10^{-4} M.

Replacing NaCl-$CaCl_2$ solutions with NaCl-$MgCl_2$ mixed ion percolating solutions decreased soil hydraulic conductivity measurably, although these effects were negligible when comparisons were made at equivalent ESP values. Ca and Mg were considered interchangeable cations under high-Na and low salt conditions.

The exact levels of exchangeable sodium and electrolyte concentration at which the hydraulic conductivity is appreciably reduced, vary with mineralogy, clay content and soil bulk density. Sensitivity to excess exchangeable sodium and low electrolyte concentration increases with clay content and bulk density. With clay percentages as low as 8 and ESP of 10 or more, the hydraulic conductivity of relatively coarse soils is appreciably reduced at electrolyte concentrations in the region of 10^{-3} M (Frenkel *et al.*, 1978).

Hydraulic conductivity reductions and soil physical properties associated with high Na (or sodic) conditions, such as clay dispersion processes, have been well documented by investigators such as Quirk and Schofield (1955), McNeal *et al.* (1968), Frenkel *et al.* (1978) and Pupisky and Shainberg (1979). In these investigations it was shown that critical or threshold concentrations dependent on the SAR (numerically equal to ESP at values below 30) governed the degree of clay particle dispersion and subsequent reductions in hydraulic conductivity.

Yousaf *et al.* (1987) concluded that soil hydraulic conductivity decreases correspondingly with clay dispersion as electrolyte concentration is decreased and SAR is increased over ranges typical of soils, i.e. 15 to 0 $mmol_c.l^{-1}$ and 0 to 20 respectively.

Russo and Bresler (1977) concluded that for practical purposes, a maximum permissible value of ESP = 15 may generally be applicable for a loam soil to maintain its structural integrity, as long as the soil solution concentration exceeds 0.01 *N*. Experiments showed that properties of the soil solution alone are not sufficient to enable characterisation of the relative hydraulic conductivities of different soils.

Alther *et al.* (1985) found that, of the aqueous solutions permeated through contaminant resistant (polymerised) bentonite and untreated bentonite, those with potassium (K^+) cations or chloride (Cl^-) anions, or both, induced the largest permeability increases with increasing electrolyte concentrations. Conversely, solutions with sodium (Na^+) cations or carbonate (CO_3^-) anions had the least impact on the permeability of bentonite. It was also observed that divalent cations have a greater initial affect on the permeability than monovalent cations. They concluded on the basis of their experimental results that the permeability of bentonite filter cake increases, over that measured with water, in response to permeation with salt solutions. The greater the concentration of electrolyte in solution, the greater the permeability increase in comparison to the water permeability. In general, it appears that there is a saturation limit for the solutions containing divalent cations in which only limited further degradation of permeability occurs beyond some electrolyte concentration. Alther *et al.* (1985) made use of the Gouy-Chapman diffuse double-layer model (described by van Olphen, 1977) to explain the permeability changes in response to permeation with electrolyte solutions. A decreasing double layer thickness consistently resulted in a more flocculated structure and a correspondingly more permeable structure. A tendency towards a more flocculated structure is caused by the decrease in the double layer thickness. Hence, an increasingly more flocculated and permeable structure would be expected with increasing aqueous salt concentration. Also, according to the model, increasing the ion valence will also cause a decrease in the double layer. The model predicts further that the smaller the hydrated ion, the closer it can approach the colloidal surface of the clay particle. Thus, all else being equal, the smaller the hydrated ion, the thinner the double layer and the greater the tendency for a more flocculated clay structure. In this regard, the data from experimental work by Alther *et al.* (1985) were inconclusive. Their data exhibit Mg and Ca in compound form with the same anion (i.e., $MgSO_4$ and $CaSO_4$, and $MgCl_2$ and $CaCl_2$). They observed $MgSO_4$ to cause a higher permeability than $CaSO_4$, while $CaCl_2$ caused a higher permeability than $MgCl_2$. These compounds differ in the size of their hydrated cation, but Alther *et al.* (1985) suggested that the anion may have an influence.

The investigation into the effects of salt on the hydraulic conductivity of a sandy soil (Pupisky and Shainberg, 1979) showed that at high ESP values and at salt concentrations > 0.01 *N* swelling is the primary mechanism responsible for hydraulic conductivity decreases. At low ESP values and at very dilute soil solutions, dispersion and clay

migration into the conducting pores are the main mechanisms for clogging pores and reducing hydraulic conductivity.

Oster *et al.* (1980) found that dispersion of both illite and montmorillonite is highly sensitive to low levels of exchangeable sodium and increases sharply with a very small increase in ESP. Whereas the flocculation value of both Ca clays is 0.25 $mol_c.m^{-3}$, the flocculation values of clays with an E_{Na} of 0.20 is 7.0 and 18.0 $mol_c.m^{-3}$ for montmorillonite and illite clays, respectively. Van der Waals attraction forces are the main forces responsible for flocculation in Ca-montmorillonite and illite systems. The edge-to-face attraction plays a dominant role in the gel formation of Na-montmorillonite. The high dispersability of the clays in the presence of a low percentage of exchangeable Na explains the high sensitivity of soils with low E_{Na} to leaching with dilute solutions and distilled water.

1.3.3 Mineralogy

Eklund (1985) conducted a study to determine the performance of two compacted native soils and the soils plus a beneficent (Volclay Bentonite Saline Seal 100) when exposed to specific paper mill waste leachate. Soil 2 was superior in its clay content, CEC, organic matter content, but had a slightly lower pH of 6.2. The clay minerals present in both soils included Ca-substituted nontronite (iron-rich montmorillonoid), kaolinite, microcline (alkali feldspar), muscovite (dioctahedral mica), quartz and dolomite. In the short term, Soil 1 had a permeability coefficient (K) of 7.4 x 10^{-8} $cm.s^{-1}$, while Soil 1 plus beneficent had a K value of <1 x 10-10 $cm.s^{-1}$. Soil 2 with and without beneficent had an initial K value of <1 x 10-10 $cm.s^{-1}$. The final permeability of Soil 1 with Waste 1 leachate after six months was 8.5 x 10^{-8} $cm.s^{-1}$. There was no evidence of shrinking or swelling, but there was indication of slight back diffusion (suspended clay in leachate above the liner). The final permeability of Soil 1 plus beneficent was 2.1 x 10^{-8} $cm.s^{-1}$. In this case major swelling had occurred as well as lateral cracking in the liner test cylinder. If swelling or cracking of the liner occurs under field conditions, failure or breaching of that liner is likely to result. Clay piping and back diffusion indicate that the clay may migrate out of the liner, weakening it as a barrier. The final permeability of Soil 2 with Waste 2 leachate was 4.4 x 10^{-9} $cm.s^{-1}$. With Waste 2 leachate, Soil 2 showed slight

shrinkage over six months. Soil 2 plus beneficent had a final permeability of $< 1 \times 10^{-10}$ $cm.s^{-1}$, and the test cylinder had developed some cracks due to swelling. Slight back diffusion of the clay into the leachate above the liner was also occurring.

Dixon (1989) indicates that a technique well suited to investigating the flocculation of clay particles is photo correlation spectroscopy (PCS). PCS can measure the hydrodynamic particle size independent of the particle shape. Dixon (1989) makes reference to Novich and Ring (1984), who used PCS to determine the critical coagulation concentration (CCC) where flocculation rate changes abruptly with salt concentration. They showed that the two lowest charged clays, kaolinite and illite, have similar CCC's.

Kaolinite is difficult to disperse because of its low negative charge. The tendency of kaolinite to flocculate may also be caused by the presence of positive crystal edge site and exchangeable cations held firmly in a Stern layer (Hunter and Alexander, 1963; cited in Dixon, 1989).

Kaolinite can be dispersed, even at neutral pH, by treatment with polyphosphate. This alters the viscosity and the positive crystal edges. Dixon (1989) illustrates the need to minimise the exposure of kaolinite to phosphate dispersants by referring to Bidwell *et al.* (1963) who showed that Al, Si and Fe were released into solution when tetrasodium pyrophosphate solutions was used to disperse kaolinite. Dixon (1989) describes how kaolinite and iron oxides (goethite, hematite and amorphous hydrated ferric hydroxides), which were originally associated, became dispersed after Golden and Dixon (1985) treated the mixture with a solution of silicate anions.

The expansive nature and negative charge of smectites cause them to be extremely reactive in soil environments. In seasonally wet and dry climates, they are responsible for most of the swelling and shrinking in soils.

Ali *et al.* (1987) concluded that the mineralogy of clay dispersed from several of the arid land soils studied was essentially independent of SAR and electrolyte concentration. All clay minerals in these soils dispersed approximately equally, with the exception of a Na-Ca saturated Bonsall soil. They concluded further that observed differences in the

aggregate stabilities are apparently not due to the differences in their mineralogies or in the differences in their relative dispersibilities of the different clay minerals. Other factors, besides clay mineralogy, appear to be more influential in controlling aggregate stability.

1.3.4 pH

Lentz *et al.* (1985) performed triaxial falling head permeability tests on specimens of kaolinite, kaolinite-bentonite mixture, and magnesium montmorillonite. Permeants used were hydrochloric acid with pH values of 1, 3 and 5 and water and sodium hydroxide with pH values of 9, 11 and 13. They suggested that a possible explanation for the decrease in permeability of the magnesium montmorillonite specimen using pH 13 NaOH permeant could be ion exchange, whereby the monovalent sodium cation replaces the divalent magnesium cation. This would cause expansion of the adsorbed double layer surrounding the clay particles, thereby reducing the effective void area available for permeant flow. Because the divalent cation is more strongly attracted to the exchange sites on the clay surfaces than the monovalent cations the exchange would not take place until the concentration of sodium ions increased enough at pH 13 to replace the magnesium by mass action. Further evidence suggested that some other reaction such as a precipitate forming in the voids of the magnesium montmorillonite was possibly contributing to the reduction in permeability. On testing of this hypothesis and analyzing by means of X-ray diffraction, it was found that a precipitate was forming and it consisted of magnesium hydroxide and calcium carbonate. This indicated that after the magnesium ions were replaced by ion exchange with sodium ions they combined with the hydroxide ion, forming a precipitate in the clay voids resulting in a significant reduction in the permeability.

In no case, for any of the clays or permeant pH values, did the permeability increase during the passage of six pore volumes of permeant, which, according to Lentz *et al.* (1985) indicates that no significant dissolution of clay minerals occurred. The only permeant that caused a significant change in permeability was sodium hydroxide at pH 13, which caused a reduction in the permeability of the magnesium montmorillonite by

a factor of 13. They concluded that for the conditions tested, pH, except for very high values, has little effect upon the permeability of the clays investigated.

These results contrasted with findings of other investigators. D'Appolonia (1985) (cited in Lentz *et al.*, 1985) found that when soil-bentonite backfill and slurry wall filter cake material was permeated with a 5% solution of NaOH the permeability increased by a factor of 5 to 10. Gordon and Forrest (1981) (cited in Lentz *et al.*, 1985) tested consolidated tailings dam slimes using sodium carbonate at pH 8 and 9. The pH of 9 resulted in permeability two orders of magnitude lower than when a pH 8 was used. This decrease was attributed to the effects of chemical reactions taking place in the soil.

At low pH values, montmorillonite and illites have higher CCC (critical coagulation concentration) values than kaolinites, which are flocculated in distilled water even at pH below 6 (Arora and Coleman, 1979, in Kretzschmar *et al.*, 1993). The CCC values of kaolinites, however, are highly pH dependent due to the predominance of edge charge, and at high pH values they can exceed the CCC value of montmorillonite.

Kretzschmar *et al.* (1993) found that the CCC values generally increased with pH with this pH dependence more pronounced for the untreated clays than for the NaOCl treated clays.

For the untreated clays the largest CCC increase occurred between pH 5.0 and 6.5 in $CaCl_2$ solutions, but above pH 6.5 in KCl solutions. The pH effect on the CCC can be explained by the pH-dependent charge on layer silicates, Fe-oxides and humic substances. With increasing pH, the particles in suspension become more negatively charged, resulting in stronger electrostatic repulsion forces. According to Schnitzer (1986; in Kretzschmar *et al.*, 1993) the macromolecular configuration of the humic substances in the untreated clays also changes with pH, which, in turn, may influence steric stabilization effects by adsorbed humic substances.

Dixon (1989) reported that by raising the pH with NaOH, kaolinite increases greatly in surface-charge density, with the greatest increase between about pH 8.2 and 10.9. Also, the apparent viscosity of kaolinite suspensions reaches a minimum at about pH 10.5. pH

values of 9.5 to 10.5 are commonly employed for kaolinitic dispersion, and it appears from viscosity and layer charge data that dispersion is achieved with greater ease with increasing pH. Dixon (1989) stresses that dissolution of aluminosilicate at high pH must be considered as a possible deleterious effect. Quirk and Schofield (1955) described how Schofield (1946) and Samson (1953) completely deflocculated a Na-saturated kaolin by small additions of NaOH which gradually raised the pH. The pH increase lead to the loss of edge-face positive charge which was present at the lower pH values.

Gipson (1985) carried out a permeability test on two alternative soils proposed for use in a compacted soil lining to minimise seepage losses from a phosphogypsum storage area. The permeant liquid was an acid liquor and the alternative materials considered for use in the lining included (1) on-site natural clayey soils, and (2) on-site silty sands mixed with commercial bentonite (Volclay saline seal, No. 100). The acid liquor had a pH of 2.2 and was high in calcium, calcium oxides, sodium, chloride and sulphate. Results of this investigation showed that the acid liquor permeant resulted in reduced permeabilities of the clayey soils compared with a tap water permeant. The tests on the commercial bentonite-silty sand mixtures with acid liquor permeant indicated that the permeability increased with time. Gipson's (1985) study demonstrated the need to use site specific soils as well as site specific fluids for permeability testing.

1.3.5 Specifically adsorbed anions and organic matter

Acar *et al.* (1985a) studied the effect of permeation fluids consisting of 0.1% and 100% solutions of nitrobenzene, acetone, phenol and benzene, representing a wide range of dielectric constants, on compacted kaolinite. Full saturation hydraulic conductivities were obtained in flexible-wall permeameters under continuous back-pressure, at hydraulic gradients of less than 100 and effective stresses of 69kPa (10psi). Reference hydraulic conductivity values were determined using 0.01 *N* $CaSO_4$ solution. Acar *et al.* (1985a) concluded that all tests at low concentrations of nitrobenzene, phenol, acetone and benzene indicated slight decreases in hydraulic conductivity. Absolute values of hydraulic conductivity with pure solutions increased with acetone and slightly decreased with phenol; three orders of magnitude decrease were observed with benzene and nitrobenzene. Hydraulic conductivity changes with pure organic fluids can be explained

by variations in flow characteristics initiated by differences in the surface forces of interaction due to changes in the chemical properties of the pore fluid (dielectric constant). The arrangement of particles and pore spaces or the fabric of compacted soils together with the effects of composition and interparticle forces constitute the soil structure, which significantly affects the hydraulic conductivity. The interparticle forces can be greatly affected by changes in the chemistry of the pore fluid.

Acar *et al.* (1985b) determined that permeating organic fluids through compacted kaolinite do not significantly change the distribution of different pore sizes. The mercury intrusion method was used to quantify the pore size after permeation with nitrobenzene, acetone, phenol and benzene. The mercury intrusion tests indicated that no significant changes had occurred in the size and distribution of pores in the 0.008 to 10μm range. Acar *et al.* (1985b) concluded that changes in hydraulic conductivity of compacted kaolinite are not due to redistribution of pore sizes due to changes in the forces of interaction between particles. Their results suggested that a decrease in hydraulic conductivity with benzene and nitrobenzene are due to the low solubility of these chemicals in water.

Results of the investigation by Bowders and Daniel (1987) showed that dilute organic chemicals (up to 80% by volume in an aqueous solution) have little effect on the hydraulic conductivity (permeability) of compacted specimens of kaolinite and illite-chlorite. The permeant liquids included methanol, acetic acid, heptane, trichloroethylene (TCE), and water. Methanol did not affect the hydraulic conductivity (K) until concentrations exceeded 80%, where an increase in K was attributed to the shrinking of the double layer caused by the reduced dielectric constant of methanol. Shrinking resulted in development of macropores and cracks and thus increased K. Decreases in K using dilute acetic acid solutions was attributed to the dissolution of some of the soil constituents at the influent end of the specimen followed by precipitation of these constituents toward the effluent end. The precipitates clogged the pores and effectively reduced the hydraulic conductivity. Pure acetic acid permeating through kaolinite in a rigid-wall permeameter increased the value of k through dissolution of these precipitates and subsequent formation of solution channels, which enhanced hydraulic conductivity.

Bowders and Daniel (1987) stress that dissolution tests should be conducted on any permeant liquids with a pH not near neutral.

Heptane and TCE had no effect on the hydraulic conductivity of either kaolinite or a chlorite illite soil when used at their solubility limits in water, but dramatically increased the hydraulic conductivity when used in their pure form. With dilution, methanol, heptane and TCE (neutral organics) substantially increased K for liquids with dielectric constants less than 35, but had no effect on K when dielectric constants exceeded 40.

Where water is the permeant, several factors strongly influence the permeability of clay-rich soils: grain size, the fabric or arrangement of particles, degree of saturation, void ratio, electrolyte concentration, composition and nature of adsorbed cations, and external pressure (Mitchell, 1976, in Budhu *et al.*, 1991). Where organic fluids are the permeants, the properties of the pore fluids, the chemical and mineralogical composition of the soil, and the nature of the adsorption of the permeant by the soil particles are potentially important factors. Budhu *et al.* (1991) were surprised to find that although illite and montmorillonite are very different mineralogically, they both respond similarly to different organic fluids. Montmorillonite (2:1 layer structure) is an expanding clay with a moderate layer charge. Illite (2:1 layer structure) is a nonexpanding clay with a high layer charge. Kaolinite (1:1 layer structure) is a nonexpanding clay with no layer charge.

The critical coagulation concentration (CCC) or the critical flocculation concentration (CFC) are two concepts which are widely covered in the literature (e.g. Kretzscmar *et al.*, 1993, and Frenkel *et al.*, 1992, respectively). In general terms, the CCC is the minimum concentration of an electrolyte required to cause flocculation of a colloidal suspension (Sposito, 1984, in Kretzschmar *et al.*, 1993). Kretzschmar *et al.* (1993) operationally define CCC as the Ca or K concentration in solution at which the optical density (OD) of the supernatant suspension was reduced to 50% after the 24-hour flocculation period. Frenkel *et al.* (1993) refer to Goldberg and Glaubig's (1987) definition of CFC as the minimum concentration of an electrolyte solution necessary to flocculate a suspension of clay under specified conditions of exchangeable cation composition, pH, and suspension concentration. According to Frenkel *et al.* (1993), determining the CFC is the most common approach in evaluating soil clay dispersion.

Kretzschmar *et al.* (1993) studied the effect of removal of organic substances and Fe oxides from clau on the critical coagulation concentrations (CCC) of dilute clay suspensions in $CaCl_2$ and KCl solutions. The CCC values of the dilute suspensions were determined for: (i) untreated clays, (ii) treatment of clays with NaOCL for the oxidative removal of organic substances, and (iii) clays treated with NaOCl and dithionate-citrate-bicarbonate for the removal of organic substances and reductive dissolution of Fe oxides. Mineralogy showed the clays to be dominated by kaolinite and hydroxy-Al-interlayered 2:1 minerals (HIM), with smaller amounts of gibbsite and poorly crystalline Al-substituted Fe oxides (hematite and goethite). Kretzschmar *et al.* (1993) found that previous investigators showed how additions of small amounts of non-crystalline Al and Fe oxides to kaolinite and montmorillonite suspensions decreased the CCC values, particularly above pH 6.5. Previous investigations showed that humic substances stabilised aqueous hematite and clay suspensions, and that adsorbed neutral molecules of high molecular weight (>3000) were most effective in stabilizing these suspensions. Also, the stabilising effect of humic acids on mineral suspensions may be a result of electrostatic stabilization or steric stabilization, with steric stabilisation possibly being the more important mechanism (Kretzschmar *et al.*, 1993 and references cited therein). Natural (untreated) soil clays commonly have much higher CCC values than comparable reference clays.

The S-shaped curves resulting from the optical density versus $CaCL_2$ ($mol_c.m^{-3}$) were characterised by three distinct regions (Kretzschmar *et al.*, 1993): (i) a region in which the suspensions are stable (all clay in suspension), (ii) a region of slow flocculation (slope), and (iii) a region of rapid flocculation (no clay in suspension after 24 hours). The CCC value represents approximately the inflexion point on these curves.

A marked decrease in CCC for the clays was observed after the removal of organic substances. For two of the clays the electrophoretic mobility was not clearly affected by either of the clay treatments, which suggested that there was little change in the surface charge density. Results of the study by Kretzschmar *et al.* (1993) strongly support the hypothesis that naturally occurring humic substances increase the colloidal stability of kaolinitic soil fine clays in aqueous suspensions. The flocculation behaviour (or colloidal stability) of a clay suspension can be characterised by its CCC under defined experimental conditions.

In their investigation into the effects of organic and inorganic anions on reference and soil clay critical flocculation concentrations (CFC), Frenkel *et al.* (1992) found that soil clays had a much higher CFC than the comparable reference clays in both NaCl and $CaCl_2$ systems; CFC of the reference clays increased by as much as a factor of 50 after addition of humic acid extracted from one of the soils and of citrate, formate, orthophosphate, carbonate and silicate (specifically adsorbing ligands); kaolinite was approximately 10 times more sensitive to the dispersion effects of these anions than bentonite. There was thus some evidence that natural levels of humic substances could be capable of inducing a high degree of dispersion of clay particles in clay soils.

Frenkel *et al.* (1992) results suggested that the soil clays were more sensitive to dispersion than their reference clay counterparts. One of their more pronounced results was the approximate 40-fold enhancement of the CFC of kaolinite on addition of humic acid equivalent to about 1% organic matter, and a fourfold increase of CFC of bentonite after addition of humic acid.

Columns holding mixtures of kaolinite, smectite and illite with quartz sand were saturated with Ca and then leached with 1 $mol_c . m^{-3}$ with one of the following inorganic and organic Na salts: chloride hydroxide, EDTA, silicate, citrate, formate, oxalate, hexametaphosphate, orthophosphate, tartrate, or humate. Hydraulic conductivity and clay dispersion were measured as a function of the various anions added. For all anions tested with the kaolinite clay-sand mixtures, the hydraulic conductivity increased above its original value and significant amounts of clay were observed in the effluent. Hydraulic conductivity of the smectite clay-sand mixtures decreased following the addition of the various anions, and only on the addition of citrate or hexametaphosphate anions was dispersed clay present in the effluent (concentration of smectite clay in the effluent was one order of magnitude lower than that for kaolinite). In both kaolinite and smectite clay-sand mixtures the hydraulic conductivity started to increase once maximum clay concentration had been reached in the effluent. The hydraulic behaviour of the illite clay-sand mixtures were similar to that of kaolinite, but their dispersive behaviour was intermediate between kaolinite and smectite.

Partial blocking of pores occurred through dispersion of smectitic clay particles after leaching with citrate and hexametaphosphate, and a sharp reduction in hydraulic conductivity followed. Thereafter, the hydraulic conductivity rapidly increased as more leachate permeated the column and purged the column of dispersed clay. Frenkel *et al.* (1992) concluded that, in the presence of small amounts of anions, of the clays studied, kaolinite was the most sensitive to dispersion. This was probably to kaolinites highest ratio of positively charged edge surface to negatively charged planar surface resulting in a high adsorption capacity. Frenkel *et al.* (1992) concluded further that there is some doubt as to the quantitative importance of clay mineral composition in the dispersion behaviour of soils.

Colloidal soil clay properties are found to be different from the specimen clays, according to Heil and Sposito (1993). They showed that flocculation at a given soluble bivalent cation charge fraction increased as the organic C content of the soil colloids decreased. Ca was more effective than Mg as a flocculant at the same soluble or exchangeable bivalent cation charge fraction. An increase in the pH caused an increase in the flocculation of one of the soils, but this trend was reversed after the removal of organic matter by H_2O_2 treatment. The amount of electrolyte required for flocculation, the effect of pH, and the relative effectiveness of Ca versus Mg on the dispersive behaviour of illitic soil all depend on the soil organic matter content.

Making use of the electrophoretic properties of an illitic soil, Heil and Sposito (1993) were able to show that electrophoretic mobility was independent of both Ca versus Mg and organic matter content. This suggested that the influence of these factors on flocculation was caused by steric effects. A possible explanation was that the adsorbed organic matter results in steric repulsion, essentially by reducing the effectiveness of the attractive van der Waals forces between mineral particles The effect of pH on the soil flocculation and the electrophoretic mobility in the presence or absence of organic matter, on the other hand, was consistent with an electrostatic mechanism. The change in net particle charge from pH 6 to 8 became more positive as the Ca concentration was increased, and this result was modelled in terms of competitive binding of H^+ and Ca^{2+} by soil functional groups.

Shanmuganathan and Oades (1983) found that clay dispersion increased when various organic and inorganic anion solutions were permeated through a kaolinitic soil. Three groups of anions were distinguished on the basis for effective dispersion and flocculation properties which also corresponded to the amount of anion adsorbed onto the soil: phosphate and fulvate > citrate, oxalate, silicate and tartrate > salicylate, catechol, aspartate, lactate and acetate.

Dispersion is undesirable and results in poor soil physical condition. Dispersion can also play a role in soil genesis by formation of a B horizon through illuviation of dispersed clay particles. The mobility of dispersed clay particles can lead to their loss from the soil profile with drainage water, particularly in the more sandy soils.

Gu and Doner (1993) concluded that negatively charged humic acid and soil polysaccharides are not flocculating agents, but they are effective dispersing agents for Na-clays and soils. The presence of soil organic matter on clay surfaces largely explained why the soil colloids were much more dispersive than the reference clays. On the other hand, the presence of polyvalent cations is necessary to facilitate the attachment of the polyanions to clay surfaces and to bridge the clay particles together, forming stable aggregates and increasing soil hydraulic conductivity. Gu and Doner (1993) concluded further that in the absence of polyvalent cations such as Al, Fe, Ca and Mg, humic substances in soil are unlikely to play a role in soil aggregate stabilization. Soil weathering that results in the release of polyvalent cations and biological activity that produces organic matter, each contribute to the formation of stable soil aggregates. The aggregates of oxisols rich in both sesquioxides and organic matter are known to be very stable and tolerant against structural deterioration (El-Swaify, 1980). On the other hand, stable soil aggregates are usually not formed in alkaline or sodic soils. This later phenomenon may also be due to the presence of humic substances as well as Na^+. Gu and Doner (1993) explained, in terms of the Marion-Babcock equation (Sposito, 1989, pp 226-245), that colloidal humic substances favour clay dispersion and migration because an electrical conductivity of 4 $dS.m^{-1}$ is equivalent to an ionic strength 58 $mol.m^{-3}$, which should be sufficient to destabilize a suspension of clay minerals in the absence of humic substances. This indicated that soil organic components (humic acid, polysaccharide,

etc.), as additional factors influencing soil clay dispersion and flocculation, should be considered in the investigation of soil stability and aggregation.

1.4 Discussion and conclusions

The literature presents supporting evidence that physicochemical interactions between soil particles, specifically clay particles, and the pore fluid does have an influence on the hydraulic conductivity of soil materials. The various methods of measuring saturated hydraulic conductivity are governed by: (i) the availability of equipment; (ii) the nature of the soil; (iii) the kind of samples available; (iv) the skills and knowledge of the experimenter; (v) the soil water suction range to be covered (if unsaturated conditions prevail); and (vi) the purpose for which the measurements are being made. The experimenter has, among others, the choice of using a flexible-wall or a fixed-wall permeameter, but must be fully aware of the advantages and disadvantages of each type, such as sidewall seepage which can produce false permeability readings.

The determination of the critical coagulation or flocculation concentration (CCC or CFC, respectively) is the most common approach taken by investigators when studying the dispersion-flocculation behaviour of soils. There is conclusive evidence in support of the concept of an electric double layer surrounding clay particles. The Gouy-Chapman Theory on the diffuse electric double layer has found wide application in the interpretation of results from dispersion-flocculation experiments (e.g. Frenkel *et al.*, 1992; Acar *et al.*, 1985a; McNeal *et al.*, 1968; Quirk and Schofield, 1955). It has been shown in the work presented that the hydraulic conductivity is reduced in soils which are subjected to fluids high in Na concentration and low in electrolyte concentration. This effect is particularly noticeable with an exchangeable sodium percentage > 15%, and is more pronounced in the expanding 2:1 clay minerals such as Na-montmorillonite and Na-bentonite.

The strongest evidence of the effect of pH on the hydraulic conductivity of soils comes from work by Quirk and Schofield (1955) and Kretzschmar *et al.* (1993). The flocculation behaviour of kaolinite is highly sensitive to pH because of the pH-dependence of

predominantly edge charge. At a high pH the CCC of a Na-saturated kaolin can exceed that of montmorillonite.

It can be concluded from the literature, that, like exchangeable cations, specifically adsorbed anions and organic substances can influence the dispersion and flocculating behaviour of soils. There is evidence that adsorbed macromolecular humic substances are effective in steric stabilization of colloidal suspensions (Kretzschmar *et al.*, 1993). An interesting outcome of work by Frenkel *et al.*, (1992) is that kaolin is very sensitive to dispersion in the presence of small amounts of anions. They concluded that there is some doubt as to the quantitative importance of mineral composition in the dispersive behaviour of soils. The main effect organic fluids have is to significantly increase permeability with organic fluids in water exceeding concentrations of 80%. This increase in permeability is attributed to the shrinking of the double layer caused by reduced dielectric constants.

The literature thus reveals the influences which soil solution chemistry has on the hydraulic conductivity of soils by affecting interaction forces between particles. However, there seems to be limited literature available on the dispersive behaviour of sandy soils. Research on the influence of landfill leachate on the dispersive properties of sandy soils in the presence of small quantities of clay minerals is urgently needed. This would have particular application where disposal sites have been established with no clay liner but have been constructed on naturally occurring sandy soils, such as those soils found along coastal plains.

Chapter 2

Soil characterisation and geochemical analysis of landfill leachate

2.1 Introduction

The previous chapter gave an account of the factors influencing the hydraulic conductivity of soils. Since the Cape Flats sand and a local landfill leachate were to be used in the column work for hydraulic conductivity measurements, they needed to be chemically and physically characterised. This chapter describes the use of various analytical procedures to meet this objective. The information gained from analyses will be used to interpret and explain the hydraulic behaviour of the soil before and after being subjected to various treatments in leaching columns.

The soil from the Cape Flats has previously been classified as a calcareous, medium sand containing negligible quantities of organic matter (Nowicki and Fey, 1994). According to Rogers (1980), borehole studies on the cenozoic sediments between Cape Town and Elands Bay showed the soils on the Strandfontein coastline to be poorly sorted, quartzose, medium sand, about 5 m deep and containing up to 26 % shell fragments. The soils represent a layer of calcareous sand blown onshore from the False Bay beaches by southeasterly summer gales.

Not only is it necessary to minimize the penetration of landfill leachate to the groundwater, but it is essential that knowledge of the geochemical composition of the leachate is available to assist in the evaluation of environmental impacts, in the event that groundwater pollution by leachate does occur. Heavy metals, for example Zn, Cr and Pb, as well as organic pollutants, can be associated with the colloidal phase of landfill leachate (Gounaris *et al.*, 1993). The implications of these associations are increased mobility of pollutants, accumulation to toxic levels in the environment over a long period of time, and their possible subsequent release through changes in ambient conditions such as temperature, pH and redox potentials. Certain inorganic compounds, such as ferric-ferrous iron hydroxides, may possess a dark colour (Schwertmann and Taylor, 1989), and their possible presence also needs to be investigated.

The possibility that sulfides, in addition to organic matter, impart to the leachate its characteristic dark colour needs to be investigated in the light of available literature. Sulfide, which is commonly found in landfill leachate as a result of reduction of sulphate, is removed from leachate bulk solution by forming precipitates with iron and heavy metals, such as lead and zinc. Due to the low solubility of these precipitates, sulfide is normally depleted in the leachate liquid phase and does not reach significant levels (Chian and De Walle, 1976; Freeze and Cherry, 1979; Kmet and McGinley, 1982; all cited in Gounaris, 1993).

2.2 Materials and methods

2.2.1 Soil sample collection

A pneumatically driven vibro-corer was used to take a core sample of the Cape Flats sand to a depth of approximately 4 m at an undisturbed site at the Coastal Park landfill. Figure 2.1 indicates the location of the Coastal Park landfill relative to the False Bay coastline and the Cape Peninsula. The soil sampling point is marked as a cross in Figure 2.1.

The vibro-corer is a useful apparatus for core sampling and investigations of stratified sedimentary materials (Martens, Davies, Baxter and Meadows, 1994; Adams and Meadows, 1994). The design of the corer is based on a tripod system and uses a portable fuel-powered generator to generate the necessary vibration through a heavy duty cable attached to a 5 m length of aluminium tubing. The sampling site was chosen about 100 m from the waste filling area to ensure the corer was penetrating an undisturbed sand profile where no previous waste tipping had taken place. The core was sampled using a 5 m length of aluminium tubing with an 8 cm internal diameter. Coring was difficult because conditions were too dry and the sand imposed a high frictional force on the walls of the tube, making penetration increasingly difficult with depth. Water was poured down the outside walls of the tube in an attempt to alleviate frictional forces and facilitate penetration. Ideally, the corer should be used for sampling saturated or very moist sediments with a high clay or mud fraction. Under ideal conditions, the very high frequency vibrations of the tube cause the sediment material in contact with the tube

walls to become fluidised, allowing easy penetration. The core sample was assumed to be undisturbed with respect to its bulk density. A 12 kg bulk sample was taken from the surface at the same location as the core sample, for later use in the leaching column work described in Chapter 3.

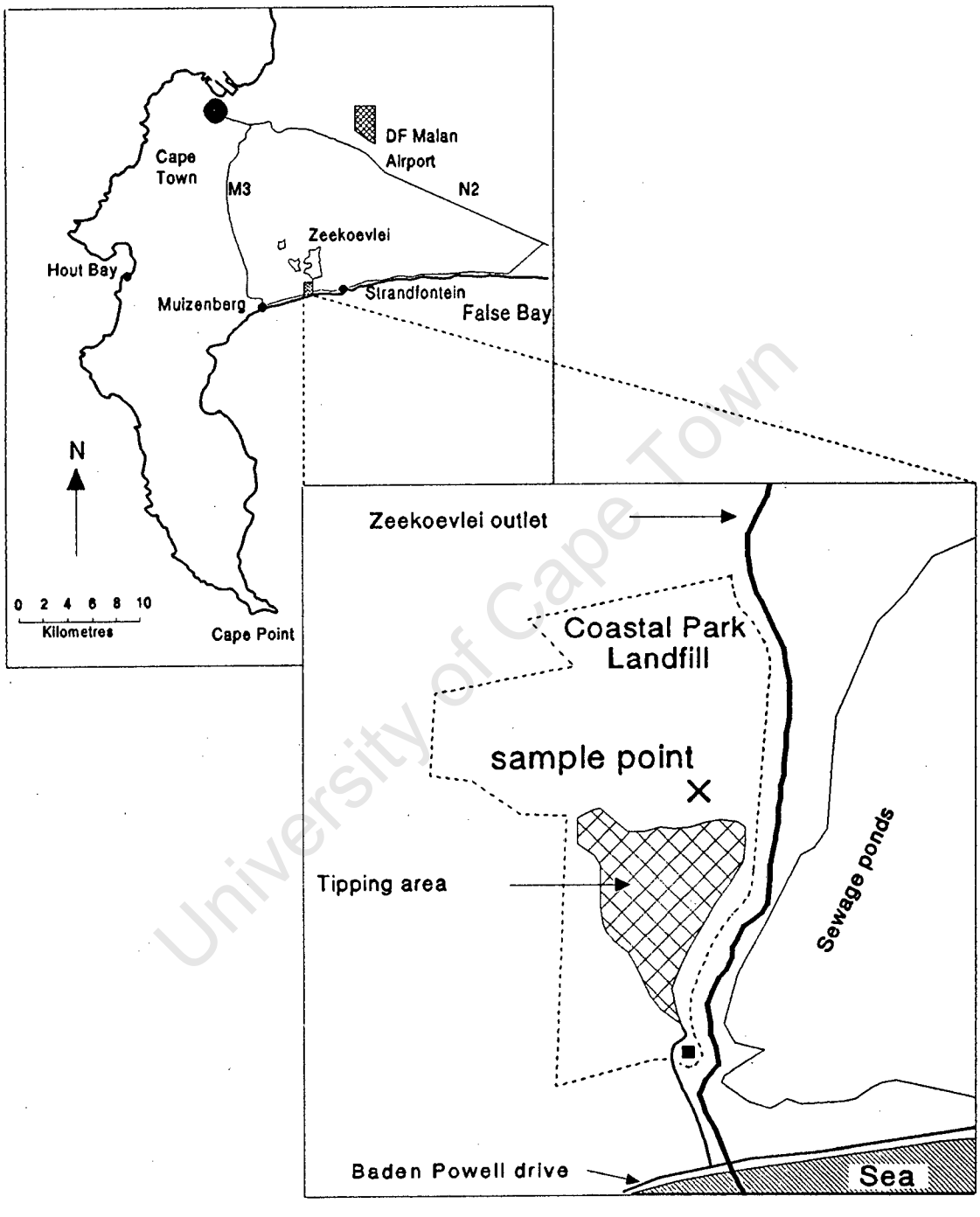

Figure 2.1 Location of the Coastal Park landfill relative to the False Bay coastline and the Cape Peninsula.

2.2.2 Soil characterisation

The particle size distribution of the sand was established by dispersion and sedimentation, in which clay (< 0.002 mm) and silt (0.002 - 0.053 mm) fractions were determined by the pipette method (Gee and Bauder, 1986), and the sand fraction by sieving.

Organic carbon content was determined by the Walkley-Black wet oxidation method (Greenberg, Trussel and Clesceri, 1985).

A 20 g sample of air dried sand was crushed for three minutes in a carbon-steel swingmill and five 3 g subsamples were used to determine an average calcium carbonate content using the Karbonat-Bombe apparatus (Birch, 1981). In this method, an aliquot of soil is reacted with conc. HCl and the pressure generated by CO_2 liberation is measured. A calibration line is constructed by analyzing 0.3, 0.5, 0.8 and 1.0 g of pure $CaCO_3$.

The dry bulk density of the sand was calculated after cutting an 80 mm length segment with a 73 mm internal diameter from the base of a 1 m-deep core previously sampled with the vibro-corer from the site. The soil was dried at 110°C and weighed. Once dried the soil was poured back into the segment of aluminium tubing from which it had originated; only a few gentle taps on the tube wall with a spatula were necessary to reduce the sand to its original volume. The dried soil weighed 543.8 g, occupied a volume of 320.6 cm^3 and thus had a bulk density of 1.696 $g.cm^{-3}$. The practical significance of this is that, in constructing leaching columns for hydraulic conductivity measurements, a realistic bulk density for this particular soil can be achieved using a disturbed soil with a minimal need for compaction.

Mineralogical analysis of the clay fraction was performed using X-ray diffraction analysis (XRD). A 1 kg sample of sand was dispersed in a 0.02 M Na_2CO_3 solution and the supernatant collected after allowing the > 2 μm fraction to settle out. The suspension was flocculated by addition of NaCl, dewatered by centrifugation, and then dialysed against distilled water to remove excess salt. A 2 cm^3 aliquot of the isolated suspension was placed on a glass slide, dried at room temperature, and analyzed by XRD. The slide was scanned with an automatic Philips PW 1390 X-Ray diffractometer, fitted with a

PW1316/9 Goniometer, a PW1771/00 Tube Tower, a PW 1050/80 radiation shield, a PW1390 channel control, a PW1394 motor control, and a PW1386/55 automatic divergence slit. CoKα radiation was used at the following settings:

1) 45 kV and 40 mA
2) NaI scintillation counter and pulse height selector
3) step scan with counting time 2 seconds and step-size of 0.05° 2-theta
4) scans form 8-40° 2-theta were used.

2.2.3 Leachate sample collection

A bulk sample of landfill leachate was collected from the Vissershok landfill (35 km NW of Cape Town) which is managed by Waste-tech (Pty) Ltd. The landfill receives hazardous liquid waste as well as general waste and is classified as an H:H waste site, i.e. a containment landfill which accepts hazardous waste with hazardous ratings 1, 2, 3 and 4 and thus required by law to adhere to certain design and construction specifications (DWAF, 1994). Leachate management at the site includes the collection and containment of leachate in three collection sumps situated on the perimeter of the waste pile. A sample was taken from each of these three sumps to make up a bulk sample contained in two 25 litre barrels, each fitted with a release tap at the base to facilitate later transfers in the laboratory. Protective clothing and a gas mask were required during sampling to minimize contact with noxious liquid and vapour.

2.2.4 Leachate analysis

The leachate was analyzed for chemical oxygen demand (COD), total Kjeldahl nitrogen (TKN), pH and electrical conductivity (EC) following standard methods (Greenberg, Trussel and Clesceri, 1985). The Council for Scientific and Industrial Research (CSIR) provides Vissershok with comprehensive leachate analyses. Included in Section 2.3.2 is one of these analyses from August 1995, which will be assumed to be representative of the typical chemical make-up of the leachate at the time of sampling.

2.2.5 Analysis of leachate solid phase

The leachate has a characteristically very dark green to black colour and has clearly visible black suspended particles. It also has a strong, nauseating odour. If left standing in a beaker in a fumehood overnight, it develops a grey-brown layer on its surface. Some of the layer material was collected for XRD analysis. When the leachate was filtered through a Whatman 45 filter paper, the solid residue turned from black to grey-brown, suggesting oxidation. This was confirmed when the original leachate rapidly changed colour from black-green to a pale orange-brown when treated with 30% H_2O_2.

The rapid change in the leachate colour upon oxidation prompted further investigation into the nature of the suspended solids. It was found that centrifugation at 6000 rpm for 60 minutes was the most practical means of separating the suspended material from the bulk fluid. Centrifugation yielded about 400 mg of suspended material per litre of leachate. Sufficient quantity of the suspended material was collected for analysis by X-ray diffractometry, X-ray fluorescence spectrometry (XRFS) and loss on ignition (LOI) analyses. A portion of the centrifuged sediment, in the form of a smooth black paste, was smeared on a glass slide with a spatula, dried under vacuum in a desiccator at room temperature in order to minimize oxidation, and analyzed by XRD as described in Section 2.2.2.

The remainder of the centrifuged sediment was frozen in plastic centrifuge tubes by immersion in liquid nitrogen, then freeze-dried in a Manifold Freeze-dryer Model B66 under a 100 kPa vacuum at −60°C over a three day period. Freeze-drying produced a loose, black, soot-like powder, confirming the suggestion (M.V. Fey - personal communication) that the technique can be used for non-oxidative drying of reduced materials. A sample of the powder was back-filled into the cavity of a rectangular frame placed above a sheet of Whatman 45 filter paper, pressed with a flat glass plate to produce a self supporting powder mount in order to minimize preferred particle orientation, and analyzed by XRD as described in Section 2.2.2. The powder was then removed, dried at 110 °C in an oven, gently ground in an agate mortar, remounted as described above, and again analyzed by XRD. This procedure was repeated after heating to 200, 400, 600, and 950 °C.

The remaining freeze-dried leachate powder was prepared as duplicate fusion discs, according to the method described by Norrish and Hutton (1969), and as a single powder briquette (according to the method of Duncan *et al.*, 1984) for analysis by wavelength dispersive X-ray fluorescence spectrometry (XRFS) using a Philips X'Unique II Spectrometer. Prior to preparation of the fusion discs, the LOI was determined gravimetrically after drying at 110 °C for 4 hours and then igniting overnight at 950 °C. Due to lack of sample, the two fusion diks were made with 0.3 g sample (disk A) and 0.1 g sample (disk B). For the powder briquette, 0.8 g sample (dried at 110°C) was first mixed with 3.2 g silica powder, again because of shortage of sample, to provided thickness for trace element determination. The briquette was prepared using 4 drops of a 2% Mowiol N 78-88 solution (Farbwerke Hoechst AG.) as a binder and pressed under 10 tonnes. Appendix I contains the instumental parameters and data quality employed in major and trace elemental determinations by XRF.

2.3 Results and discussion

2.3.1 Soil characterisation

The soil characterisation data are presented in Table 2.1. The soil can be classified as a calcareous medium sand (see textural triangle in Appendix II), with predominantly medium (42%) and fine (41%) sand and 16% coarse sand (Table 2.1). Light microscopy indicated shell fragments, especially in the medium and coarse sand fractions. The organic matter and clay contents of the soil are negligible. The dry bulk density of the soil was found to be 1.70 $g.cm^{-3}$.

The XRD analysis (Fig. 2.2) showed the fine fraction (<0.002 mm) of the soil to be predominantly calcite (d-spacing = 3.04 Å) and quartz (d-spacing = 3.34 Å). The very weak aragonite peak relative to calcite, does not necessarily indicate a low concentration of aragonite. Van de Spuy and Willis (1991) showed that low intensity aragonite peaks could still indicate the presence of significant quantities of aragonite. They demonstrated, by XRD of coal using CuKα radiation, how the presence of aragonite peaks could be missed on XRD scans due to overlapping or masking by other peaks of more abundant minerals, such as quartz. Aragonite is commonly associated with fresh shell fragments

and some was identifiable with a d-spacing of 3.40 Å (Doner and Lynn, 1989). The presence of calcite/aragonite was confirmed by HCl treatment and repeating the XRD analysis. The sand contained 12.8% $CaCO_3$ (Table 2.1).

Table 2.1 Particle size distribution, organic carbon and calcium carbonate content of the soil sample.

Constituent	%
Coarse sand	16.3
Medium sand	41.5
Fine sand	40.7
Very fine sand	0.7
Coarse silt	0.07
Fine silt	<0.01
Clay	<0.01
Organic carbon	<0.01
Calcium carbonate	12.8
Dry bulk density ($g.cm^{-3}$)	1.70

Aragonite is biologically precipitated as shell fragments and not through processes of dissolution and reprecipitation which is responsible for the formation of calcite coatings on sand grains. The summer southeasterly gales is the most likely mechanism for the arrival of shell fragments from the beach. Subsequent relocation within the soil through dissolution and reprecipitation, because of the moderately low solubility of $CaCO_3$ produces carbonate coatings on the quartz grains (Doner and Lynn, 1989). In this study carbonate coatings on particles of the sand were observed under the light microscope. This was confirmed by reaction with HCl which removed the coatings, causing effervescence to occur in the process.

The occurrence of calcium carbonate in soils is heavily dependent on the climatic conditions. Generally, carbonates remain in the soil because of the aridity of the climate where rainfall is insufficient to cause loss of the carbonates from the soil by leaching. Under these conditions secondary carbonates accumulate near the surface (Doner and Lynn, 1989). During the rainy winter season at Coastal Park there may be some removal

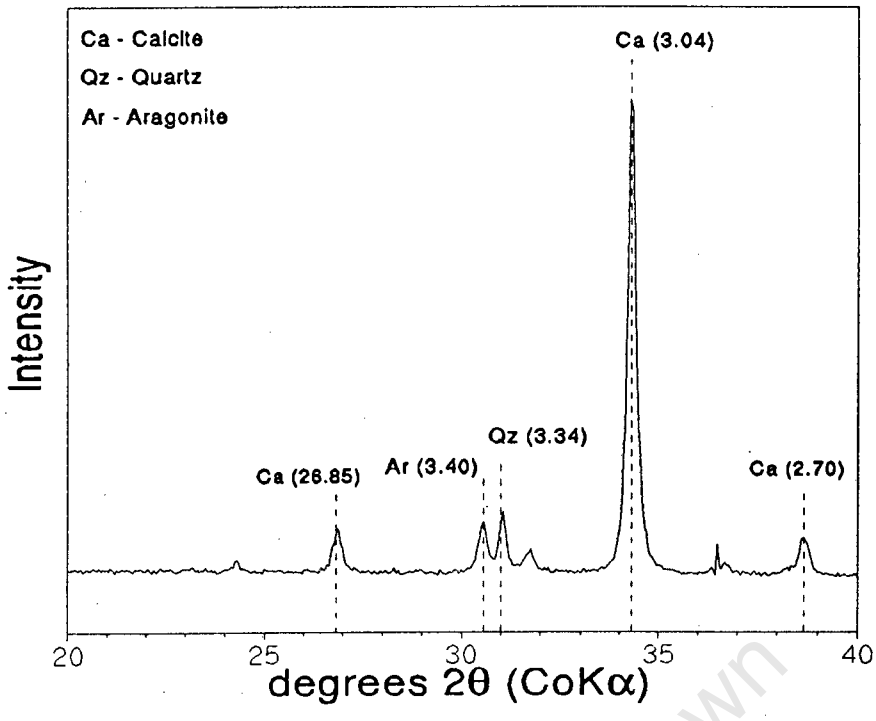

Figure 2.2 X-ray diffraction pattern of <0.002 mm fraction of Cape Flats soil sampled from Coastal Park landfill site. Values in parentheses are d-spacings in Å units. Unlabelled peaks are secondary peaks.

from the upper layers and deposition in the deeper layers, but with minimal loss from the soil profile through leaching. Continued carbonate precipitation may force H_2O to move laterally by closing soil pores through the formation of continuous carbonate coatings between neighbouring soil particles. Such layers in a soil matrix may form in layers where H_2O movement is significantly impeded (Stuart and Dixon, 1973; cited in Doner and Lynn, 1989).

The particle size distribution determines the amount of carbonate necessary to plug the soil fabric. Generally, the more gravel, the less carbonate is required to form a continuous layer. Also, the narrowing of pores by growth of calcite precipitate coatings on quartz particles could enhance the filtering or straining effects of the sand grains, causing progressively more blocked pores to develop and increasingly impeding H_2O movement.

The sandy soil lacks clay minerals and organic matter and thus a poor nutrient status would be expected. However, any nutrient depletion may be offset by the capacity of calcite to adsorb significant quantities of phosphates (Doner and Lynn, 1989) and nitrates

(Jurinak and Griffin, 1972). Depending on their age, phosphates may be present as precipitates of dicalcium and octacalcium phosphates (Freeman and Rowell, 1981; cited in Doner and Lynn, 1989). These phosphate macrolayers are thought to form at low P concentrations (Cole *et al.*, 1953).

2.3.2 Leachate composition

2.3.2.1 Liquid phase

The dark green colour of the liquid phase and the characteristic change to a yellow-brown colour upon oxidation fits descriptions by Schwertmann and Taylor (1989) of green-blue mixed Fe^{2+} - Fe^{3+} hydroxides known as *green rusts*. Feitnecht and Keller (1950), cited in Schwertmann and Taylor (1989), recognised green rusts as layer-structured double-hydroxy salts which, as shown by Taylor (1973), are members of the pyroaurite ($Mg_6^{2+}Fe_2^{3+}(OH)_{16}CO_3 \cdot 4H_2O$) group of compounds. These compounds contain divalent and trivalent cations which are octahedrally coordinated with OH in hexagonally close-packed octahedral sheets identical to those in $Fe(OH)_2$. The positive charge in these sheets, because of the presence of Fe^{3+} ions, is balanced by anions such as CO_3^{2-}, Cl^- and SO_4^{2-} which occupy the interlayer region between octahedral sheets. The Fe^{2+}:Fe^{3+} ratio is generally in the range of 2-to-4. The green rusts are capable of undergoing isomorphous substitution with Al (Taylor and McKenzie, 1980; in Schwertmann and Taylor, 1989).

Green rusts have not yet been identified in soils, probably because of the rapidity with which oxidation causes structural breakdown and colour change to yellow-brown (Bernal *et al.*, 1959; cited in Schwertmann and Taylor, 1989). Stable green rusts can be synthesized easily in the laboratory where they have been shown to play an intermediate role in the formation pathways of other iron oxides. The composition of the green rust, particularly the type of interlayer anion (Cl, SO_4, CO_3), the Al for Fe substitution and the rate and environment of oxidation, are probably the major factors influencing lepidocrocite:goethite ratios in soils (Taylor and McKenzie, 1980; Taylor, 1980; cited in Schwertmann and Taylor, 1989). Because of the similarity in original colour, the instability on exposure to air, and the products of oxidation, these compounds have been

suggested to be those responsible for the dark green colours in reductomorphic soils (Taylor and McKenzie, 1980; cited in Schwertmann and Taylor, 1989). This information suggests the possible presence of green rusts in the landfill leachate at Vissershok.

No comprehensive chemical analysis of the liquid phase was conducted on the Vissershok bulk sample. However, Table 2.2 contains recent results of an analysis of the Vissershok leachate conducted by the CSIR in August 1995. The analysis is assumed to be of the solution phase. Included in the table are COD, TKN, EC and pH determinations of the leachate used in this study. The values determined in this study all fall within the range of values for samples 6622 and 6623, indicating chemical similarity between the leachate used in this study and leachate analyzed previously at Vissershok.

Table 2.2 Chemical analysis of Vissershok landfill leachate solution phase.

Lab No.	6622 *	6623 *	This study
Chemical oxygen demand ($g.dm^{-3}$)	45.8	9.9	13.0
Total Kjeldahl nitrogen ($mg.dm^{-3}$)	2280	860	955
Ammonia as N ($mg.dm^{-3}$)	2030	730	-
Conductivity @ 25°C ($ms.m^{-1}$)	4200	1700	2680
pH	6.9	7.8	7.7
Metals ($mg.dm^{-3}$)			
Cd	0.10	<0.05	-
Cu	0.32	0.12	-
Cr	0.10	<0.05	-
Ni	1.4	0.43	-
Pb	0.8	0.3	-
Zn	2.9	0.3	-

* Conducted by CSIR on 15-09-95

2.3.2.2 Solid phase

Tables 2.3 and 2.4 list the results of the XRF analyses for major and trace elements, respectively. Relatively high concentrations of heavy metals, including Zn, Cu, Ni, Co and Mn, and to a lesser extent, V and Pb, were found in the in the solid phase of a landfill leachate studied by Gounaris *et al.* (1993). Compared to the solution concentrations in Table 2.2, Cu, Cr, Ni, Pb, and Zn have relatively high concentrations in the solid phase (Table 2.4). Gounaris *et al.* (1993) describe the relationship between dissolved and colloidal (solid phase) species by means of the distribution coefficient (K_d), defined as the ratio of the solid phase concentration of the metal to its concentration in the dissolved phase.

Table 2.3 Major elemental concentrations (expressed as oxides) in the solid phase of Vissershok leachate (by XRFS using fusion discs).

Constituent	Concentration (%)
SiO_2	4.08
TiO_2	0.07
Al_2O_3	1.86
$^{*}Fe_2O_3$	18.68
MnO	0.13
MgO	0.67
CaO	4.19
Na_2O	6.65
K_2O	0.98
P_2O_5	1.39
H_2O	3.82
LOI	57.26
TOTAL	101.12

*Total Fe expressed as Fe_2O_3

Table 2.4 Elemental concentrations in the solid phase of Vissershok leachate (by XRFS using powder briquettes).

Trace element	Concentration ($mg.kg^{-1}$, except S)
S	19.5 (%)
Zn	2470
Cu	225
Ni	427
Co	121
Mn	1030
Cr	121
V	28
Mo	2.5
Nb	0.2
Zr	5.2
Y	2.8
Sr	30
U	0.3
Rb	4.7
Th	0.4
Pb	16

To allow for comparison, the log K_d values determined by Gounaris *et al.* (1993) for specific colloidal size fractions are presented as a range in Table 2.5, together with log K_d values determined for similar metals in this study. The Vissershok leachate K_d values were calculated using solution metal concentrations contained in Table 2.2 and solid phase metal concentrations from Table 2.4.

Table 2.5 Calculated log K_d values for the leachate solid phase based on heavy metal concentrations contained in Tables 2.2 and 2.4 for the Vissershok leachate (pH 7.66).

Metal	Solution phase	Solid phase	log K_d	
	($mg.l^{-1}$)	($mg.kg^{-1}$)	This study	*Literature
Cu	0.32	224	2.85	-
Cr	0.10	121	3.08	2.68 - 3.17
Ni	1.40	427	2.48	-
Pb	0.80	16	1.30	1.99 - 3.20
Zn	2.90	2469	2.93	2.58 - 4.24

* Range reported by Gounaris *et al.* (1993)

The K_d values in Table 2.5 suggest that, generally, heavy metals in the leachate have a high affinity for the colloidal fraction. It should be noted, however, that some of these metals may be present as precipitates, for example sulfides. The significance of this is that if the colloids do become mobile in the environment, for instance through deep-seepage to groundwater, then there is a risk of heavy metal pollution, particularly where the colloids may encounter pH and pE gradients which may cause the release of heavy metals back into solution from the solid phase (Sposito, 1984).

The colloids in the Vissershok landfill leachate formed a stable suspension in a glass beaker, with the larger particles settling out of suspension over a 48 hour period. Colloidal mobility during flow through porous media is directly related to stability (Gounaris *et al.*, 1993). Unstable colloids tend to aggregate into larger particles and be

removed by settling or filtration. Colloidal stabilisation can result from electrostatic or lyophilic mechanisms (Shaw, 1980; Napper, 1983; Ross and Morrison, 1988; Stumm and Morgan, 1980; cited in Gounaris *et al.*, 1993). Repulsive electrostatic forces between colloids having surface charge of the same sign, preventing the particles from approaching one another closely enough for attractive van der Waal's forces to dominate. In the case of hydrophobic colloids, effective collisions can be minimised through a process known as steric stabilisation. In this process macromolecules containing hydrophilic segments adsorb to the colloid surface. The extension of hydrophilic segments away from particle surfaces make particle aggregation thermodynamically unfavourable (Gounaris, *et al.*, 1993). Gounaris *et al.* (1993) found that lyophilic stabilization of the combined enthalpic-entropic type is the major mechanism responsible for the stability of colloids in the leachate.

The black colour of landfill leachate is usually attributed to the presence of organic material (M.V. Fey - personal communication). The leachate powder, when freeze dried, was soot-black, but when heated to 110°C it turned an orange-brown colour, indicating that oxidation had occurred spontaneously at that temperature. It was suggested earlier that the leachate suspension contains a mixture of Fe^{2+} - Fe^{3+} hydroxides known as green rusts, which impart a dark green colour to it. These green rusts can undergo a colour change from dark green to an orange-brown upon oxidation. Hematite may also be imparting some blackness to the solid phase, since this mineral also turns from a black to a brown when oxidised (Schwertmann and Taylor, 1989). Sulfides are commonly formed in landfill leachate by reduction of sulphate and are removed from the liquid phase by formation of highly insoluble sulfide precipitates with iron and heavy metals (Gounaris *et al.*, 1993). The highly reduced properties of the leachate, and a sulphur concentration of 19.5% suggests the presence of sulfides of iron and other heavy metals, in addition to the iron hydroxides. Although no pyrite was detected in the XRD analysis, it is very likely that Fe sulfide is present as either mackinawite (tetragonal, Fe_9S_8) and/or greigite (cubic, Fe_3S_4), which are both initially amorphous or poorly crystalline (thus not detected by XRD) and, with time, change to pyrite (FeS_2) under reducing conditions (Schwertmann and Taylor, 1989). The relatively high solid phase concentration of Zn (2469 $mg.kg^{-1}$), relative to the liquid phase (2.9 $mg.l^{-1}$) together with the reducing conditions and high sulphur concentration of 19.5% in the solid phase, would suggest the

possible presence of a weakly crystalline sphalerite (ZnS). It should be noted that a fraction of the sulphur is probably associated with the organic phase inherent in the leachate, and thus not only present as sulfides.

The leachate paste, separated and dried from the leachate suspension, was analyzed by XRD. Calcite was the only detectable crystalline phase. The diffraction pattern is presented in Figure 2.3 in which the dominant calcite peaks are indicated. The slide was analyzed before and after treatment with 0.01 M HCl for the removal of carbonates. Upon exposure to the atmosphere, a sample of the Vissershok leachate developed a white crystalline layer which was collected, dried and found by XRD also to consist primarily of $CaCO_3$.

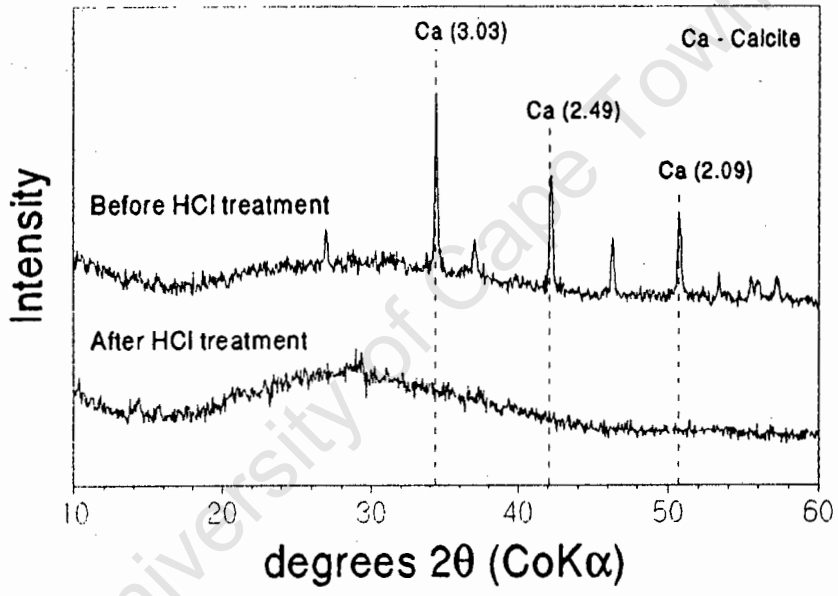

Figure 2.3 XRD diffraction pattern of leachate paste prepared as a smear on a glass slide. Analysis was carried out before and after carbonate removal by 0.01 M HCl.

The development of this layer can probably be explained in terms of calcite solubility data. According to Doner and Lynn (1989) , if the activity of calcium is high enough, then under the correct conditions of pH and partial pressure of CO_2, calcite would precipitate out of solution. The analysis of the leachate by the CSIR (Table II.1 in Appendix II) shows that calcium concentrations can be almost as high as 3200 $mg.l^{-1}$, which is almost certainly high enough for calcite to precipitate out at pH 7.7 and log

(P_{CO_2}) of 4.5. Further speciation investigations should be carried in regard to saturation status of the leachate with respect to minerals such, as calcite.

2.3.2.2.1 Freeze dried leachate solid phase

This exercise was carried out more from an interest point of view and no significant conclusions can be drawn from it since the transformations displayed here are not what could be expected under *in situ* conditions.

The freeze dried leachate solids were heat treated for 4 hours each at temperatures of 110, 200, 400, 600 and 950°C, and between each heat treatment XRD analysis was conducted. The diffraction patterns (Fig. 2.4) show how the mineralogy of the leachate solid phase undergoes alteration with increasing temperature. This is evident where a rise in temperature from 200 to 950°C sees the formation of hematite taking place and its increase in crystallinity is shown by the narrowing of the peak at d-spacing of 2.52 Å. Halite is lost from the sample between 600 and 950°C.

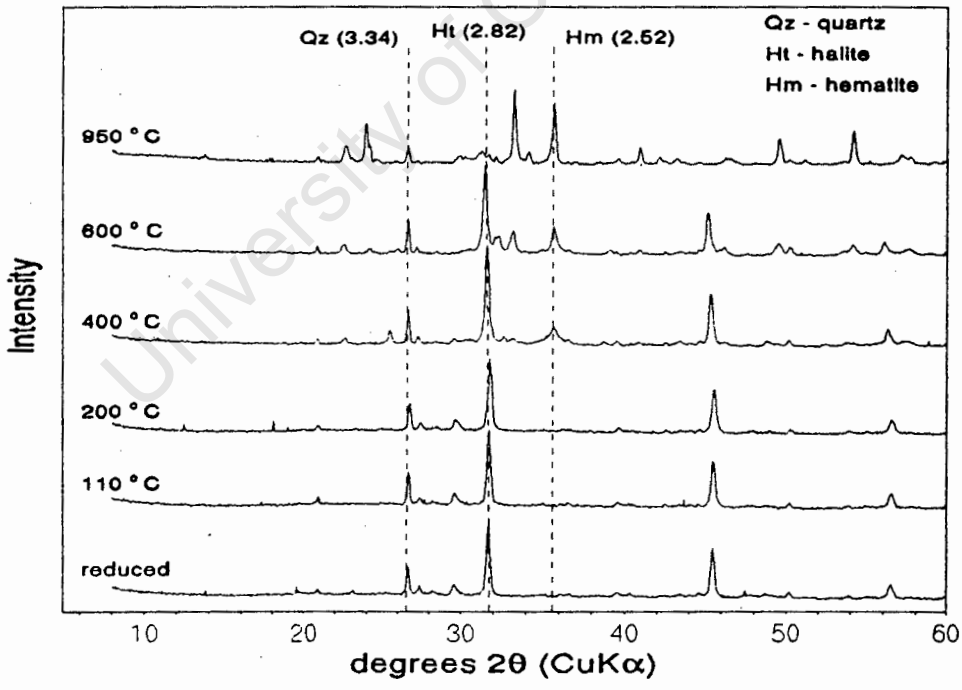

Figure 2.4 XRD analysis of freeze-dried Vissershok landfill leachate before heat treatment (ie. reduced) and at each stage of heat treatment. Unlabelled peaks are secondary to those for the three minerals. Values in parentheses are d-spacings in Å units.

In this same range quartz tends to diminish from the sample. The typical colour changes observed after each stage of heating were: (1) reduced = black; (2) 110°C = orange-brown; (3) 200-600°C = brown; and (4) 950°C = purple. The purple colour after heating to 950°C indicates the presence of macrocrystalline hematite. When this purple powder is crushed under mortar and pestle, it undergoes a colour change from purple to red, as it alters from a macro- to a microcrystalline form. If organic matter had been present in the sample, then diagnostic peaks of maghemite would have appeared in Figure 2.4 at 400°C (Schwertmann and Taylor, 1989). Time prohibited further detailed analyses of the leachate solids.

The powder mounts used in this XRD analysis showed halite to be present in the solid phase, compared to the glass slide prepared from the paste, which showed the presence of calcite in the solid phase. This can be explained in terms of presence of CO_2 and the time allowed for it to dissolve in the leachate. More CO_2 dissolved in the paste as it dried on the glass slide, combining with the Ca ions to form the $CaCO_3$. Freeze-drying precluded the CO_2 from the solution, thus preventing the opportunity for $CaCO_3$ formation.

2.4 Conclusions

The Cape Flats sand can be classified as an aeolian, calcareous, medium sand, with negligible organic carbon content and extremely clay-depleted, and poor in structure. Although unlikely to provide any significant chemical attenuation of leachate constituents, the sand does possess some buffering capacity because of the presence of carbonates. Physical parameters such as dry bulk density, organic carbon content and particle size distribution have been determined and will be considered in Chapter 3 in the context of leaching column experiments.

X-ray diffractometry and XRF techniques have shown that the leachate solid phase is enriched with iron and heavy metals. It has been proposed that, because of similarity in original colour (dark green) and instability on exposure to air, the leachate contains compounds which consist of mixed Fe^{2+} - Fe^{3+} hydroxides known as green rusts (Schwertmann and Taylor, 1989). These green rusts are members of the pyroaurite group

of compounds. It is also suggested that amorphous sulfides such as mackinawite (Fe_9S_8) and/or greigite (Fe_3S_4) are present in the leachate solid phase. Hence it is not necessarily organic matter which imparts a black colour to the leachate. Centrifugation, unless specific size fractions of the solid phase are required, has been shown to be a suitable method for isolating the leachate solid phase for purposes of XRD and XRF analyses. There are indications that the method of sample preparation for XRD analysis can produce different results, shown here where a glass slide of leachate paste produced calcite peaks, compared to halite peaks (amongst others) produced from powder mounts. The high EC (26.8 $mS.m^{-1}$) and the chemistry of the leachate liquid and solid phases are likely to influence the hydraulic conductivity of the soil materials to be tested in column experiments in Chapter 3.

Chapter 3

Factors affecting hydraulic conductivity of Coastal Park soil

3.1 Introduction

Landfills are often constructed on soils which have been physically and chemically manipulated to achieve minimum hydraulic conductivity for the containment of leachate generated by degradation processes in the waste pile. According to the locally specified *Minimum Requirements* (DWAF, 1994) the maximum acceptable hydraulic conductivity allowable for a clay liner is 1 x 10^{-7} cm.s^{-1} for a hazardous (H:H) landfill containment liner, and 1 x 10^{-6} cm.s^{-1} for general (G:L:B^{+}) classified landfill liner. Soil-lined facilities have been used extensively for the containment and disposal of waste liquids (Brown, 1986), and much work has been conducted on the physicochemical factors influencing the hydraulic conductivity of soils and the ability of various solutions, including landfill leachate, to compromise the structural integrity of soils used as liners. The liner materials must be capable of withstanding chemical attack which can physically alter their structure and enhance hydraulic conductivity. It is the clay particles which play the most important role in the efficacy of the material as a containment liner.

Van Olphen (1977) and McBride (1994) describe the mechanisms and processes, such as those in electric double layer theory, taking place at the surface of clay particles which govern their behaviour in suspensions and in soil environments. Hillel (1982) and Klute and Dirksen (1986) cover the principles of Darcy's Law, hydraulic conductivity and the methods employed for its calculation. Frenkel, Levy and Fey (1992) have investigated clay dispersion and hydraulic conductivity of clay-sand mixtures as affected by the addition of various anions. The effects of clay type (montmorillonite, kaolinite and vermiculite) and content, exchangeable sodium percentage, and electrolyte concentration on clay dispersion and soil hydraulic conductivity have been researched by Frenkel, Goertzen and Rhoades (1977). Quirk and Schofield (1955) showed how electrolyte concentration influences soil permeability. Nowicki (1994) conducted a literature review on the properties and efficacy of clay liners, with particular reference to bentonite-amended soils and the influence of landfill leachates. Eklund (1985) compared the

effects of water and waste water on the performance of landfill soil liners. Smith and Fey (1993) and Nowicki and Fey (1994) conducted laboratory studies into the chemical manipulation of soils for their use in the sealing of landfills. Yousaf, Ali and Rhoades (1987) demonstrated the effects of electrolyte concentration and sodium adsorption ratio on the cumulative dispersion of four arid land soils and their relative hydraulic conductivities.

This chapter describes an investigation of the hydraulic properties of the Cape Flats soil and how these are influenced by addition of clay amendments and leaching with various solutions, including landfill leachate. Coastal Park has been classified as a G:L:B^{+} landfill (Novella and Eichstädt, 1995) and will thus be required in future to adhere to the requirements for leachate management at the site. Leachate management includes the construction of a suitable liner below the landfill, as well as the placement of a capping system over the landfill upon its closure. At present there is no liner and the landfill has been constructed on the very permeable calcareous sand, allowing an average 2 m separation between the base of the landfill and the Cape Flats aquifer. Information on hydraulic conductivity of clay-sand mixtures to be used as possible liner materials are provided here in the event that future extensions of the landfill will incorporate a liner of some sort. It is anticipated that this type of information will have some use in the modelling and design of such a liner. From a more general perspective, some of the physicochemical factors influencing the hydraulic conductivity of the amended soil are investigated, particularly when leached with landfill leachate.

3.2 Materials and methods

Laboratory experiments using leaching columns have been conducted to investigate the influence of various amendments and leaching solutions on the hydraulic conductivity of the Coastal Park soil. It was established in Section 2.2 that, if required, a disturbed sample of soil could be used to pack a leaching column equivalent to *in situ* dry bulk density. This can be achieved by packing a known mass of soil to a known height in the column. In this way a dry bulk density of 1.70 g.cm^{-3} can be achieved.

Rigid-wall perspex columns were used in all column work, each having an internal diameter of 54 mm and total height of 175 mm. The columns were fitted with rubber stoppers, which allowed access into the top and out of the base of the column through glass tubes to which flexible polyethylene tubing had been fitted. The basal support of the columns consisted of a perspex screen covered with glass fibre tissue. In all experiments a constant head device (Mariotte bottle) was used to maintain a 1 m hydraulic head. In most cases, the hydraulic conductivity measurements were carried out in duplicate. Figure 3.1 schematically represents the experimental apparatus used in the laboratory.

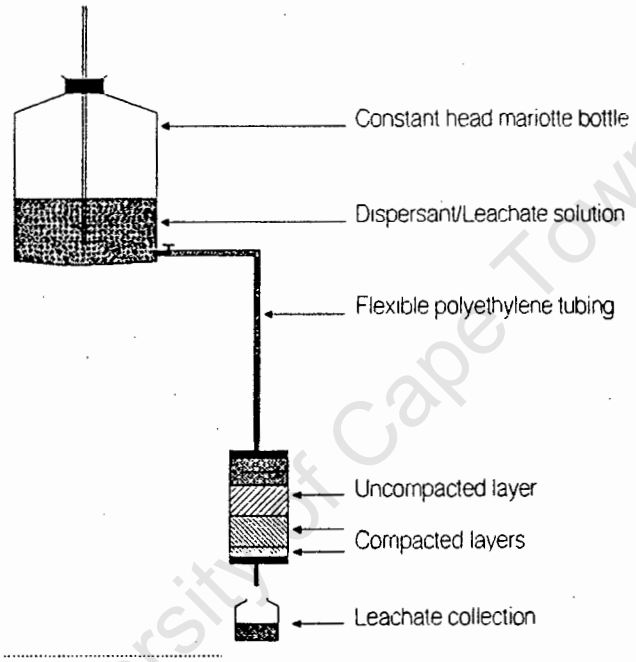

Figure 3.1 Experimental apparatus employed in all leaching column work in this study. The mariotte bottle was used to maintain a 1m hydraulic head.

The particle size distribution and organic carbon content of the soil were determined using standard soil analysis techniques (Gee and Bauder, 1986) and have been reported in Chapter 2.

3.2.1 Procedure for column packing

Two methods of packing the columns were investigated. The first was to use a soil column height of 14 cm with high-density middle and lower layers and a low-density top layer. This method showed relatively high variability in hydraulic conductivity when tested in duplicate. Degassing in the top low-density layer was evident from the

accumulation of gas bubbles. The presence of air bubbles has the effect of decreasing the cross-sectional area contributing to water movement, and their accumulation in the soil tends to reflect an artificial reduction in the measured hydraulic conductivity (Hillel, 1982).

The second method of packing was adopted from Nowicki and Fey (1994) where a soil column of approximately 5.7 cm was packed with same the layering and compaction, but smaller quantities based on mass. Nowicki and Fey's (1994) method was found to produce less variability in hydraulic conductivity measurements when tested in duplicate, and was thus used in all subsequent column work. Figure 3.2 shows the typical column packing used. The soil column packed to an average height of 5.7 cm with lower (27.7 g) and middle (101.7 g) high-density layers (compacted to 95 % dry volume), and a low-density top layer (80.2 g, uncompacted).

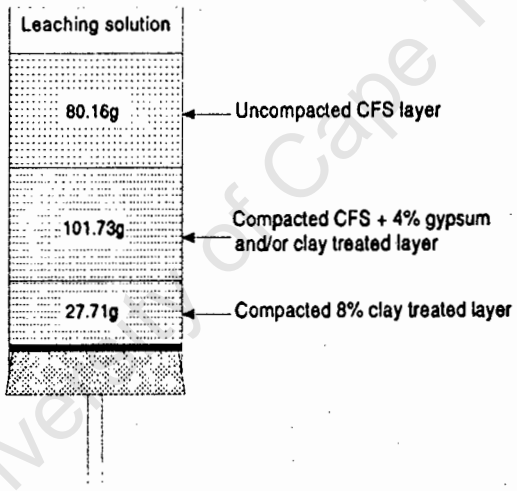

Figure 3.2 Typical soil column packing used to measure hydraulic conductivity of Cape Flats sand treated with either 8% bentonite or kaolinite and a 4% gypsum-treated middle layer (in the case of kaolinite).

3.2.2 Soil treatments

The soil (<2 mm) was air dried and either packed as an untreated soil column, or packed with a clay and/or gypsum amendment. The clay amendments were mixed into the entire soil column.

3.2.2.1 Kaolinite amendment

The first clay amendment to be used was a locally derived kaolinite, sampled in Rondebosch (south western Cape Province). A particle size distribution of this local soil had been conducted by HKS Law Gibb (G. Robertson, personal communication) and found to contain more than 40 % clay. A 10 g sample of this soil was dispersed in 50 ml 0.1 M Na_2CO_3, and a 2 ml aliquot of the suspended clay was placed on a glass slide and allowed to dry before analysis by XRD using methods already described in Section 2.2.2. The glass slide was scanned through a range of 8 to 40° 2θ using CuKα radiation. The diffraction pattern (Figure 3.3) showed the soil to consist predominantly of kaolinite (d-spacing 7.19 Å) with some illite (d-spacing 3.34 Å).

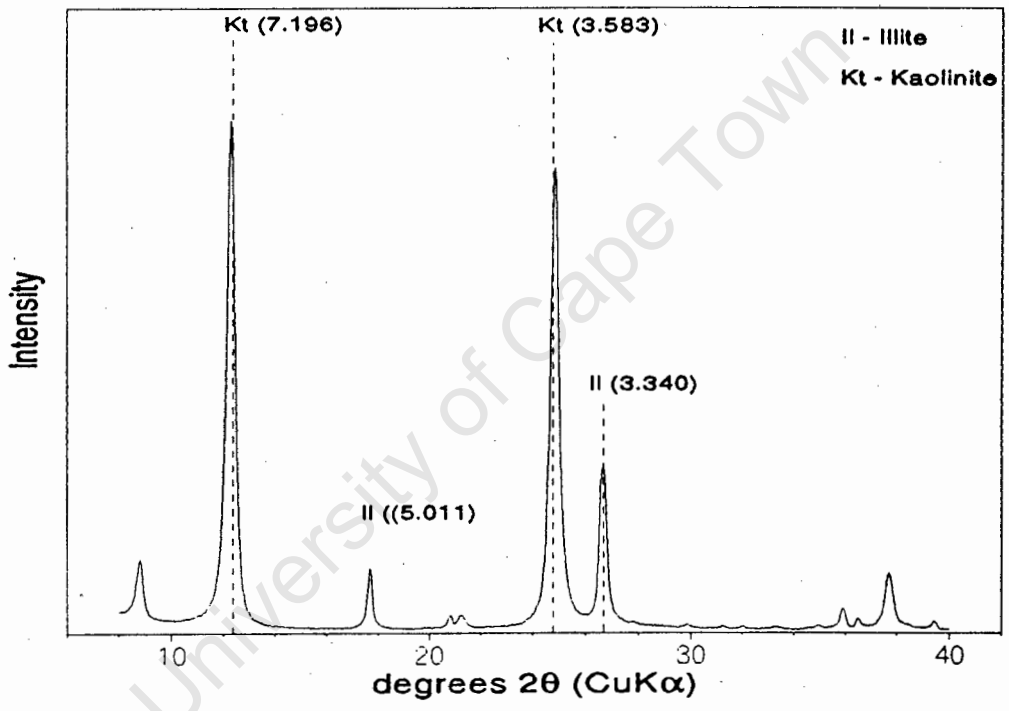

Figure 3.3 XRD analysis of the local clayey soil sampled from Rondebosch (south western Cape Province), showing kaolinite and illite peaks. The clay was used as an 8 % kaolinite amendment of the Coastal Park sand. Values in parentheses are in Å units. Unidentified peaks are secondary to those already labelled.

This local clay was manually crushed using mortar and pestle and mixed with the soil in an automatic mixer for 2 hours to produce an 8 % kaolinite/soil mixture.

3.2.2.2 Kaolinite and gypsum amendment

A second kaolinite amendment was prepared in exactly the same manner as the first except for the addition of 4 % gypsum (an optimum treatment established in previous work by Nowicki and Fey, 1994) to the middle layer. The kaolinite clay was chosen to assess the effectiveness of a non-swelling clay in sealing the soil and subsequently reducing its hydraulic conductivity to acceptably low values of less than 10^{-6} cm.s^{-1} (DWAF, 1994). The gypsum was used as a flocculant which would promote the formation of a clay seal by inducing flocculation of the clay particles previously dispersed by 0.02 M Na_2CO_3 solution (Nowicki and Fey, 1994; Smith and Fey, 1993).

3.2.2.3 Na-bentonite amendment

The third clay amendment used was a commercial Na-bentonite, which was mixed with the soil in the same manner as the kaolinite to produce an 8 % bentonite/soil mixture. An 8 % bentonite mix was found to be the optimum treatment in previous work (Nowicki and Fey, 1994), required to reduce hydraulic conductivity to values of less than 10^{-6} cm.s^{-1} (DWAF, 1994). Bentonite possesses extreme shrink-swell properties characteristic of smectitic clays and its use as a soil amendment for the formation of seals has been widely studied (Oster, Shainberg and Wood, 1980; Clem, 1984; Simons and Reuter, 1984; Brown, 1986). In this experiment, the Na-bentonite has been added to the soil and its swelling properties exploited by treatment of the columns with a dilute dispersant, 0.02 M Na_2CO_3, to form a seal before leaching with Vissershok landfill leachate. No gypsum was added to these columns, and before leaching the columns were saturated with distilled water under a 2 m hydraulic head.

A single column was also packed with soil to an equivalent *in situ* dry bulk density of 1.70 g.cm^{-3} with no clay or gypsum treatment. This column was leached with Vissershok landfill leachate in order to determine the effect the leachate would have on the hydraulic conductivity of the soil.

3.2.3 Leaching solutions

To test the effects of various soil treatments and leaching solutions on the hydraulic conductivity of the soil, the following leaching solutions were used:

1) Distilled water - Determination of hydraulic conductivity of control columns and testing their reproducibility. Also used as pretreatment of bentonite amended soil to enhance swelling and hence sealing effect of the clay before leaching with Na_2CO_3 and then leachate.

2) 0.02 M Na_2CO_3 - Dispersant solution for all clay-treated soil columns to promote formation of a clay seal through clay swelling and dispersion.

3) Vissershok landfill leachate - Leached through all columns immediately following leaching either with distilled water (in the case of the untreated columns), or 0.02M Na_2CO_3 (in the case of the clay amended soil columns), to test the efficacy of the clay amendments as seals against landfill leachate.

When the columns were leached with the landfill leachate, work was conducted in a fumehood. Volumes were collected either in small glass beakers or in 100 ml plastic bottles equipped with teflon-lined lids.

3.2.4 Calculation of hydraulic conductivity

The equation for the calculation of hydraulic conductivity of the soil columns was derived from Darcy's Law (Hillel, 1982):

$$q = \frac{K\Delta H}{h} \qquad (9)$$

also,

$$q = \frac{V}{At} \tag{10}$$

where q is the specific discharge rate (i.e., the volume of liquid flowing through a unit cross-sectional area per unit time t), V is the liquid volume (cm^3), A is the cross-sectional area (cm^2), t is the time (seconds), K is hydraulic conductivity ($cm.s^{-1}$), ΔH is the hydraulic head (cm) and h is the height of the soil column (cm). Combining equations (1) and (2) and rearranging:

$$K = \frac{V}{At} \cdot \frac{h}{\Delta H} \tag{11}$$

Experimentally, only volume (V) and time (t) need to be measured for the calculation of hydraulic conductivity, since A, ΔH and h are constant. The log K values (y-axis) are plotted against the number of pore volumes (x-axis, unitless). The pore volume of the soil column was calculated by subtracting the volume occupied by the sand particles from the total soil volume. The particle density of the soil was assumed to be equivalent to quartz (2.67 $g.cm^{-3}$), allowing the volume of particles to be calculated using the mass (209.6 g) of soil in the column. Total volume was calculated using internal radius of the column (2.7 cm) and average height of soil (5.6 cm). The average pore volume was found to be 49.8 cm^3.

3.3 Results and discussion

3.3.1 Estimation of *in situ* hydraulic conductivity

Appendix III contains the measurements of time and volume and the calculated hydraulic conductivity values. The hydraulic conductivity K of approximately 3 x 10^{-2} $cm.s^{-1}$, determined during the test for reproducibility using distilled water (Figure 3.4), can be considered equivalent to the *in situ* hydraulic conductivity of the soil found at Coastal Park, because the assumption has been made that the bulk densities are equivalent. The columns showed a narrow range of K values and minimal fluctuation. The value of 3 x 10^{-3} $cm.s^{-1}$ reported by Blight, Ball and Vorster (1994) is an order of magnitude lower, most likely due to their value been determined by pump and recharge tests during

geohydrological investigations which contrasts with the laboratory scale work in this study. Dunn (1984) points out that even with careful laboratory testing procedures, there is still an inherent variation between laboratory and field conditions, and this can have large effects on hydraulic conductivity. A review of the state of practice in laboratory measurement of hydraulic conductivity of "saturated" soils by Dunn (1984) indicated that few standard test methods are available and the extensive degree of variation in the test procedures used in practice can result in measurements of saturated hydraulic conductivity varying over two or three orders of magnitude for a given soil.

3.3.2 Effect of landfill leachate on soil hydraulic conductivity

Figure 3.5 shows how the K value of the untreated soil can be affected when leached with landfill leachate. The soil in the column was packed to the *in situ* dry bulk density found at Coastal Park. The K value underwent a significant reduction of almost three orders of magnitude, ranging form a maximum of $10^{-1.5}$ to a minimum of $10^{-4.5}$ $cm.s^{-1}$. Visual inspection of the column revealed that the suspended particles in the column were accumulating on the surface of the soil and becoming entrapped in the pores of the soil matrix. Blocking of pores was observed to take place at the surface and to gradually migrate downward as a dark band which moved half-way down the soil column after the passage of about 45 pore volumes. The K value stabilized to some degree after 35 pore volumes. The collected effluent was centrifuged at 6000 rpm for 60 minutes forcing separation of suspended solids from the bulk solution, showing that the finer colloidal fraction was capable of passing through the soil without being filtered out. This may be significant if heavy metals are concentrated in the colloidal fraction (Gounaris *et al.*, 1993), and are thus capable of entering the environment through mobilisation of these colloids.

3.3.3 Effect of kaolinite on soil hydraulic conductivity

The 8 % kaolinite treatment of the soil on its own did not maintain a reduced hydraulic conductivity (Figure 3.6). Initial leaching of the columns with a 0.02 M Na_2CO_3 dispersant solution reduced the K value from about $10^{-3.6}$ $cm.s^{-1}$ to a minimum of $10^{-4.5}$ $cm.s^{-1}$.

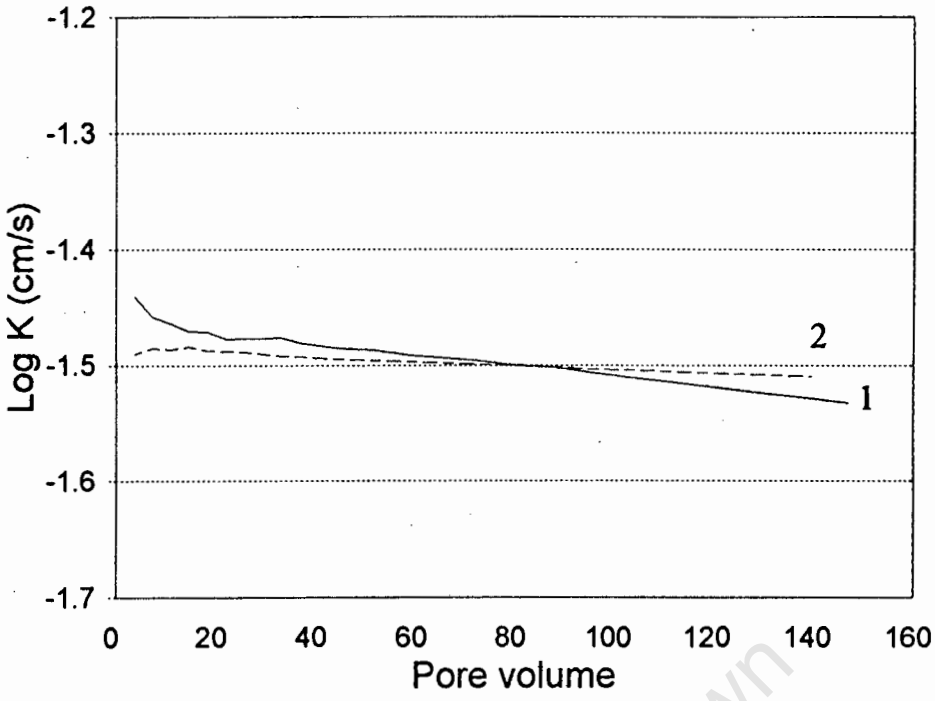

Figure 3.4 Hydraulic conductivity as a function of pore volume for duplicate columns (1 and 2) packed with a known mass of Cape Flats sand to a known height, and leached with distilled water using a constant 1m hydraulic head.

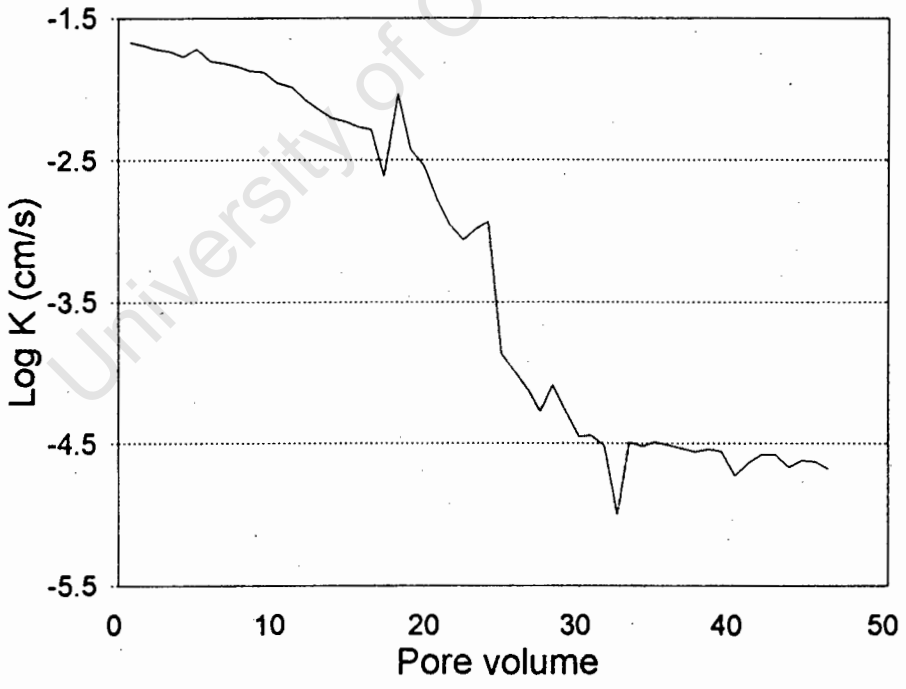

Figure 3.5 Change in hydraulic conductivity K (cm/s) with passage of landfill leachate through a single column of Cape Flats sand packed to original dry bulk density.

Clay was purged from the column which coincided with a rise in K value from $10^{-4.5}$ to about $10^{-3.2}$ cm.s^{-1}. At this stage landfill leachate was introduced and the column showed some stability with minor variation of K values between $10^{-3.1}$ and $10^{-3.4}$ cm.s^{-1}.

The fact that leachate had little effect on the hydraulic conductivity, and that a rise in hydraulic conductivity was observed while still leaching with the dispersant, suggests that significant amounts of clay particles were purged from the column. The result shows that too much dispersion could become self-defeating, and, that by the time leachate passes through the column, any sealing effect that the kaolinite might have had is lost. The presence of kaolinite in the effluent as a result of purging is consistent with the findings of Frenkel, Levy and Fey (1992). They found that kaolinite dispersed easily in the presence of small concentrations of anions. In this study, carbonate anions were added to the kaolinite/soil mix as 0.02 M Na_2CO_3 solution. These anions have the effect of satisfying the positive edge charge on the kaolinite silicate crystals which provides them with an excess negative charge (van Olphen, 1977). Consequently, the edge-to-face mode of attraction and flocculation in kaolinite (Schofield and Samson, 1954; cited in Frenkel *et al.*, 1992) is eliminated. The interaction between the edges and the planar surfaces of the kaolinite particles is now controlled by the diffuse double-layer forces and depends on the type of exchangeable cation and electrolyte concentration (Frenkel *et al.*, 1992).

It is interesting to note that the minimum K value achieved with untreated soil was about $10^{-4.6}$ cm.s^{-1} after the passage of 50 pore volumes, in contrast with the minimum of $10^{-3.7}$ cm.s^{-1} achieved with 8% kaolinite after the same number of pore volumes. In the latter case, purging of kaolinite out of the soil column increased the pore volume and enhanced the hydraulic conductivity as a result (water flow is proportional to the fourth power of the radius of soil pores, according to Hillel, 1982). In the absence of clay treatment no significant purging of soil particles occurred and blocking of pores by leachate suspended material effectively reduced the pore volume and hence hydraulic conductivity.

3.3.3 Effect of kaolinite and gypsum on soil hydraulic conductivity

Figure 3.7 demonstrates how 8 % kaolinite in conjunction with 4 % gypsum-treated middle layer was more effective in sustaining a reduced hydraulic conductivity than the

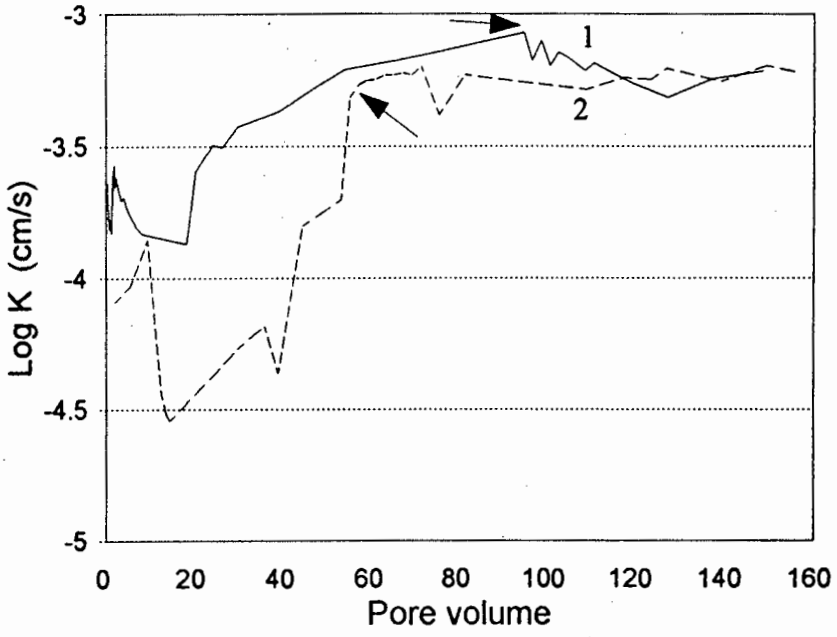

Figure 3.6 Change in hydraulic conductivity K (cm/s) with passage of solution through duplicate columns (1 and 2) of Cape Flats sand amended with 8% kaolinite. Arrows point to the stage at which leachate was introduced after pretreatment with 0.02M sodium carbonate.

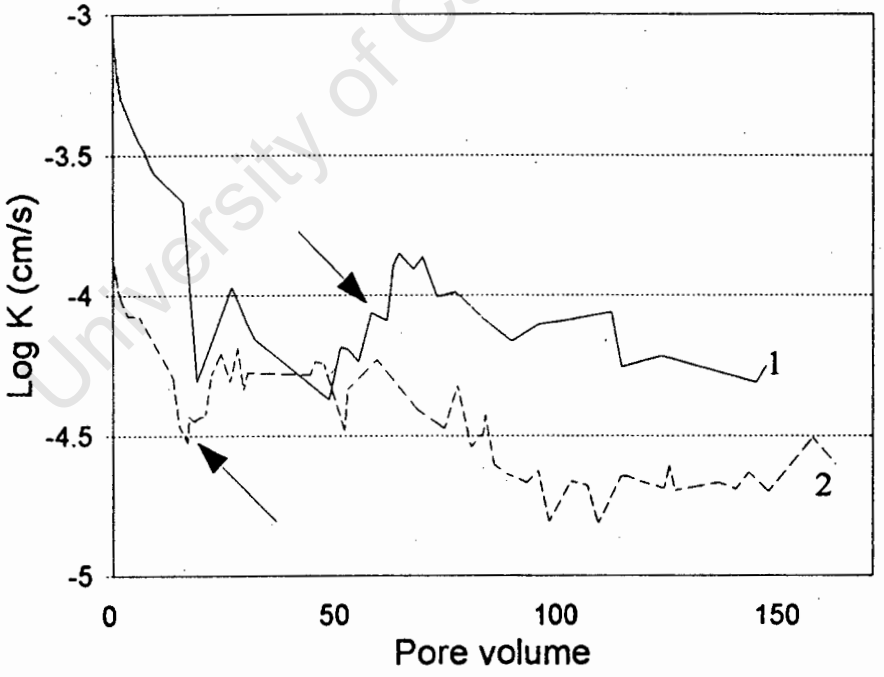

Figure 3.7 Change in hydraulic conductivity K (cm/s) with passage of solution through duplicate columns (1 and 2) of Cape Flats sand amended with 8% kaolinite throughout and 4% gypsum in the middle layer. Arrows point to the stage at which leachate was introduced into the columns after pretreatment with 0.02M sodium carbonate.

8 % kaolinite treatment on its own. The purpose of the gypsum was to induce flocculation of the dispersed clay to promote the formation of a seal layer at a particular depth in the soil column. The K values show an initial drop from 10^{-3} to $10^{-4.5}$ cm.s^{-1} when leached with 0.02 M Na_2CO_3. With introduction of leachate, the K value showed a slight increase and then showed some stability, fluctuating between values of about $10^{-3.9}$ and $10^{-4.7}$ cm.s^{-1}.

3.3.4 Effect of Na-bentonite on soil hydraulic conductivity

Hydraulic conductivity of the soil showed greatest response, compared to the other treatments, to the 8 % bentonite treatment (Figure 3.8). The minimum initial K value of $10^{-7.8}$ cm.s^{-1} (1.6×10^{-8} cm.s^{-1}) was achieved by leaching the columns with the 0.02 M Na_2CO_3 dispersant solution. However, a steep increase in K occurred to a maximum value of about $10^{-4.7}$ cm.s^{-1} when leached with landfill leachate. The columns show some degree of stability at approximately 10^{-5} cm.s^{-1} after about 8 pore volumes. The initially low hydraulic conductivity showed how effective the bentonite could be in sealing the pores of the sandy soil. However, the inherently high EC (26.9 mS.cm^{-1}) of the landfill leachate, and thus its correspondingly high ionic strength, resulted in a fairly rapid shrinking of the bentonite clay. As shrinkage occurred, the darkness of the leachate revealed where side-wall seepage had developed as a result of the soil column pulling away from the inner walls of the permeameter. The steep increase in hydraulic conductivity was attributed to this side-wall seepage. Cracking was also observed, however, at the interface between the top low-density layer and the middle high-density layer (Figure 3.9). A high osmotic potential gradient existed between the landfill leachate and the interlayer water molecules of hydration, because of the high EC of the leachate. As a result, the leachate may have had a dehydrating effect on the soil column by causing water molecules to migrate from their interlayer positions into the bulk leachate solution (McBride, 1994, p287 *et seq.*). The shrinking action of the soil column and the cracking were partially attributed to this osmotic potential gradient. Alther (1987) also found that increasing valency and electrolyte concentration enhanced the permeability of bentonite, because of shrinkage and cracking. The high Ca^{2+} concentration in the leachate (Table 2.3) may contributes further to the shrinking process by replacing exchangeable Na^+ and counteracting the Na-induced swelling.

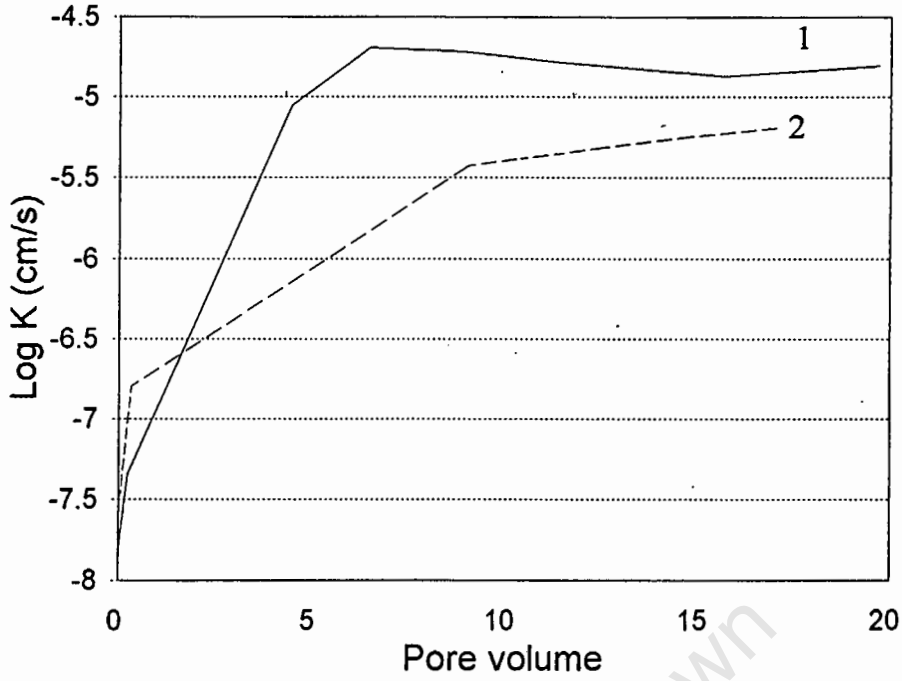

Figure 3.8 Change in hydraulic conductivity K (cm/s) with passage of solution through duplicate columns (1 and 2) of Cape Flats sand amended with 8% Na-bentonite throughout.

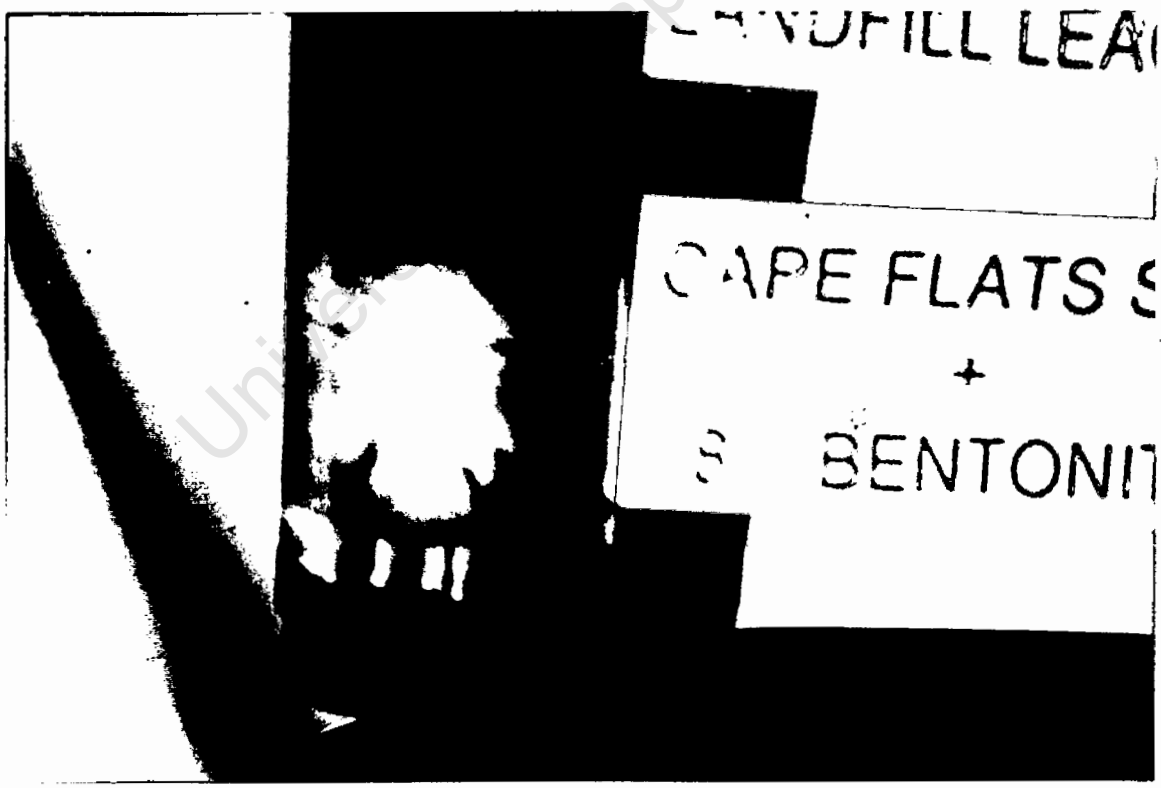

Figure 3.9 Photograph of one of the 8% bentonite treated soil columns showing the position of salinity-induced cracking at the low-density/high-density interface in the soil. The darkness toward the outer edges of the column is evidence of severe side-wall seepage

What is significant about the results presented in Figure 3.8, is that the enhanced hydraulic conductivity as a result of salinity-induced shrinkage could materialise on a field scale. A bentonite-modified liner could be ineffective in containing landfill leachate because its integrity could be undermined by cracking, induced by the high EC of the leachate.

3.3.4.1 Pure bentonite versus bentonite-sand as a potential liner

Although experimental work precluded the test of pure bentonite as a potential liner, the following discussion attempts to outline the comparative efficacy of a pure bentonite liner versus that of a bentonite/soil mixture (M.V. Fey, personal communication).

Consider a column of 8 % bentonite-amended soil, and a second, of pure bentonite. The sand particles and shell fragments can be assumed to be entirely impermeable to the leaching fluid. If the bentonite has reached full swelling capacity by saturation with, say, a dilute Na_2CO_3 solution, then the bentonite column will possess a higher hydraulic conductivity relative to the bentonite/soil mixture. This is because the cross-sectional area contributing to permeability is higher in the case of pure bentonite than the bentonite-sand mixture where a fraction of the cross-sectional area consists of impermeable quartz and shell particles (Figure 3.10).

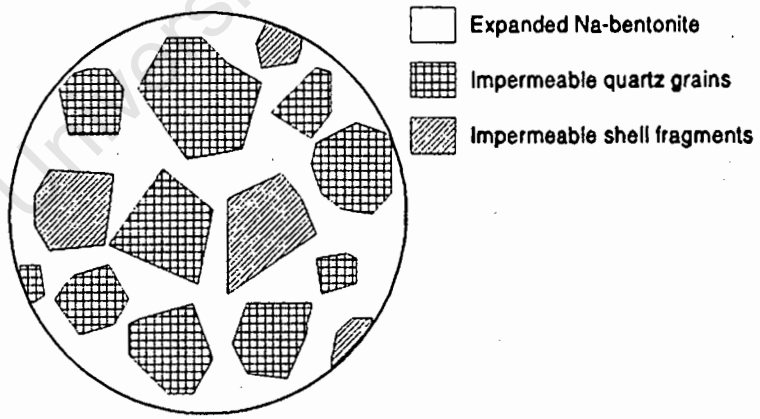

Figure 3.10 Cross-section of a hypothetical bentonite-amended soil column. The quartz grains and shell fragments are effectively impermeable. Only the expanded clay fraction contributes to fluid flow.

3.4 Conclusions

This study has shown that the laboratory determination of *in situ* hydraulic conductivity of the Coastal Park soil using leaching columns is an order of magnitude higher than that determined previously by others in the field by pump and recharge tests during hydrogeological investigations. However, this variation is attributed to problems of scale and differences in methods.

Passing landfill leachate through the soil, which had been packed to *in situ* dry bulk density, resulted in significant reductions in hydraulic conductivity of almost three orders of magnitude. This may be attributed to suspended solids being filtered out by the sand, causing a subsequent reduction in pore space and hydraulic conductivity. Although filtering of these suspended solids did take place, the finer colloidal fraction passed through the soil column. This may be significant in the case where no clay liner is constructed below a landfill, since potentially hazardous metals such as zinc, chromium, vanadium and lead, concentrated in this high-surface area colloidal fraction through processes of adsorption and precipitation (Gounaris *et al.*, 1993), may escape into the environment.

An 8 % kaolinite amendment failed to maintain a reduced hydraulic conductivity when ponded with landfill leachate after prior dispersion with Na_2CO_3. Clay dispersion and its subsequent purging from the column occurred, possibly because too much dispersant was applied, destroying any desirable sealing effect of the kaolinite.

The 8 % kaolinite amendment of the sand was most effective in sustaining a reduced hydraulic conductivity if 4 % gypsum was added to the middle layer in the column. Further tests would be required, however, to test the period of time over which the integrity of the seal can withstand treatment with leachate.

Of all the clay amendments, 8 % bentonite resulted in the lowest hydraulic conductivity values. However, it was not possible to maintain these low values once landfill leachate had been introduced, because of enhanced K values as a result of shrinking (leading to cracking and side-wall seepage) induced by the high EC of the leachate compared with

the prewetted clay. This behaviour could also be expected to manifest itself under field conditions.

Chapter 4

Modelling the water balance and leachate generation at the Coastal Park Landfill using LEACHM

4.1 Introduction

The impact that waste disposal sites have on groundwater resources is an area of serious concern (Parsons and Jolly, 1995) and preservation of these resources is of critical importance if the future water demand of this country is to be satisfied. The control and management of leachate generated at any landfill site will be determined by the climatic conditions and the hydrogeological response of the landfill. The portion of precipitation that remains after runoff and evapotranspiration will infiltrate the waste-pile. There, the water extracts or dissolves suspended materials from the waste-pile, forming leachate and causing potential contamination of the groundwater. Because of its importance when investigating the risk of groundwater contamination, an estimate of the rate and temporal distribution of leachate generation needs to be obtained.

A number of techniques and guidelines are available for water balance calculations to indirectly estimate leachate generation at waste disposal sites (Blight, 1995; DWAF, 1994; Blight, Ball and Vorster, 1994; Gee, 1981; Perrier and Gibson, 1980; Fenn *et al.*, 1975; Dass *et al.*, 1977). In the water balance method, the amount of water percolating through the solid waste is obtained by subtracting the runoff, change in soil moisture content, and evapotranspiration from total precipitation. DWAF (1994) uses this technique in southern Africa to evaluate the suitability of sites for the establishment of landfills.

Parsons (1995) conducted a literature survey in order to compare and assess recharge estimation and approaches and techniques for predicting leachate generation. The Waste Site Water Balance (WSWB) method is used to predict the volume of leachate generated at waste disposal sites, to determine co-disposal (solid / liquid) ratios and to specify site design and engineering requirements. Groundwater recharge evaluations are also used to predict the generation of landfill leachate at waste disposal sites.

Parsons (1995) states that results obtained using the WSWB method are not compatible with knowledge concerning groundwater recharge. This is particularly true for arid and semi-arid conditions. From a geohydrological point of view it is difficult to quantify groundwater recharge, even though it is relatively simple to show that recharge does occur throughout the country. It has become widely accepted that in areas where the annual evapotranspiration rate exceeds the annual precipitation rate (ie. water deficit areas), water contamination due to landfill leachate is not a problem. This suggests that leachate only poses a threat to groundwater in about 20% of South Africa (Parsons, 1995). However, processes of recharge and leachate generation are so significantly different that the two do not equate. It would therefore be technically invalid to use recharge estimation techniques to predict the volume of leachate produced by a landfill. Parsons (1995) concluded that the WSWB method is invalid under arid and semi-arid conditions during comparative recharge estimation studies and, further, that the widespread application of the WSWB method to estimate the volume of leachate generated at a waste pile should be terminated. Because no alternative means of estimating the leachate volume generated is available, a conservative strategy must be adopted when siting, designing and operating a landfill.

The water balance technique also has some drawbacks. In the computation of leachate volume, the moisture storage capacity of the waste body and the actual process of moisture movement through the refuse or soil layers are not taken into account. Furthermore, the technique does not include the computation of leachate mound head, which is generated as a result of leachate accumulation at the bottom of the landfill (Ahmed *et al.*, 1990).

In Minimum Requirements legislation (DWAF, 1994), the Climatic Water Balance, used to classify landfill sites as either leachate generating or not, is calculated using only two climatic components of the full water balance, namely rainfall and A-pan or S-pan evaporation. The simple calculation can be represented as follows:

$$B = R - E \qquad (12)$$

where B is the Climatic Water Balance (mm), R is rainfall (mm) and E is the evaporation (mm) from a soil surface. The value of B is calculated for the wettest year in the rainfall record, and then calculated for successively drier years. This calculation is repeated until it is established whether (DWAF, 1994):

a) B is *positive* for *less than one year in five* for the years for which data are available. In this instance, no significant leachate generation will occur on account of the climate, and the site is classified B-. If the Minimum Requirements for the siting, design, and operation are met and only dry waste is disposed of, no leachate management should be necessary at B- sites; or

b) B is *positive* for *more than one year in five* for the years for which data is available. In this instance, *significant leachate generation* will occur and the site is classified B+. Leachate management is a Minimum Requirement at B+ sites.

The modified Darcian equation used by Knight (1983; cited in Parsons, 1995) appears to provide a realistic tool for estimating leachate production. However, it is still to be tested and verified using appropriate field data from a wide range of climatic, hydrological and site design conditions. The one-dimensional FLOW model reported by Blight *et al.* (1994) also shows promise, but again appropriate validation is required (Parsons, 1995). The Hydrological Evaluation of Landfill Performance (HELP) model developed by Schroeder *et al.* (1984) is a quasi-two-dimensional deterministic model that computes leachate generation in a quasi-steady-state flow condition. Blight *et al.* (1994) used observed leachate generation data from experimental cells at Coastal Park to calibrate FLOW for modelling the effects of water flow at landfills, and compared its predictions with those of the HELP model. The two data sets did not agree since the HELP model was calibrated by means of observations made in Wisconsin, in northern USA (Blight *et al.*, 1994). It is relevant to point out, however, that for the time period involved, FLOW successfully predicted leachate generation.

Ahmed *et al.* (1990) developed a two-dimensional model for the estimation of leachate flow through landfills which included computation of surface runoff, evapotranspiration, and infiltration at the upper boundary of the landfill. Their model took into account both

saturated and unsaturated flow of leachate, and involved modifications of Darcy's equation. In this model, equations of flow are solved through an iterative process using finite difference techniques.

Up to now, the literature apparently contains no mention of modelling landfills in their post-operative condition. In particular, attempts at modelling the dynamic moisture balance of a closed landfill, with a suitable capping system and established vegetation under ambient climatic conditions, seems to have been neglected. This chapter describes how the LEACHM model may be used to predict the water balance and leachate generation at Coastal Park, if a hypothetical landfill cover consisting of Cape Flats sand is constructed. The aim was to establish an optimal soil depth and vegetation cover required to maintain a deficit water balance in a hypothetical soil cover. This modelling exercise was based on the rationale that leachate generation can be kept to a minimum, or even eliminated, if the capping system is maintained in a deficit moisture condition. The results of the simulations have not been verified against any field data, and the predictions have been based on a set of stated assumptions. LEACHM is regarded as a sophisticated model and is well suited for modelling the movement of water in saturated and partially saturated soil conditions (Prof. J.H. Moolman, University of Stellenbosch, personal communication). It can be run on a stand-alone computer, and is accompanied by a software manual. LEACHM is particularly well suited to the Coastal Park situation because it makes use of the Richards equation, which is applied assuming horizontally homogeneous soils, ie. soils which do not have strong structure or preferential flow paths such as cracks or worm holes. In this study, where a cover of Cape Flats sand is proposed as a cover at Coastal Park, this assumption is valid.

4.2 LEACHM model concept

LEACHM (Leaching Estimation And CHemistry Model) Version 3, developed by Hutson and Wagenet (1992), is a process-based simulation model of water and solute movement, transformations, plant uptake and chemical reactions in the unsaturated zone. It is suitable for unsaturated or partially unsaturated soils to a depth of about 2 m. It consists of five submodels arranged in a modular basis: LEACHC describes transient movement of inorganic ions; LEACHB describes microbial population dynamics in the presence of

a single growth-supporting substrate; LEACHN describes nitrogen transport and transformation; LEACHP simulates pesticide displacement and degradation; and LEACHW describes the water regime. LEACHW can be considered the engine block of LEACHM since the other submodels rely on the soil moisture status and water movement in the profile. LEACHW makes use of the finite difference form of Richard's equation as a means of predicting water contents, fluxes and potentials. It relies on the input of soil physical parameters, weekly A-pan evaporation (mm) and daily rainfall (mm) data. The profile is divided into a number of horizontal segments of equal thickness with nodes designated in the centre of each segment where mass balancing is performed. Included are two additional boundary nodes, one above the surface and one below the lowest depth, which are used to maintain boundary conditions. The form of Richards equation used to derive transient vertical flow and which is derived from a combination of Darcy's law and the continuity equation (Hillel, 1980), is:

$$\frac{\partial \theta}{\partial t} = \frac{\partial}{\partial z}\left[K(\theta)\frac{\partial H}{\partial z}\right] - U(z,t) \qquad (13)$$

where θ is the volumetric water content per unit volume of soil ($m^3.m^{-3}$), H is the hydraulic head (mm), K is the hydraulic conductivity ($mm.d^{-1}$), z is the depth (positive in the downward direction, in mm) and U is the sink term accounting for the water lost per unit time by transpiration. One problem with the above equation is that it contains two dependent variables (θ and H). Thus, by defining the differential water capacity, $C(\theta)$, as

$$C(\theta) = \frac{\partial \theta}{\partial h} \qquad (14)$$

where h is the soil water pressure head (mm), the Richards equation can be written in the form

$$\frac{\partial h}{\partial t} C(\theta) = \frac{\partial}{\partial z}\left[K(\theta)\frac{\partial H}{\partial z}\right] - U(z,t) \qquad (15)$$

in which the pressure potential, ($\partial h/\partial t$), is the only dependent variable (Hutson and Wagenet, 1992).

The retentivity data required to solve the Richards equation are estimated from the texture and bulk density of the soil horizon, using the method of Campbell (1974) as modified by Hutson and Cass (1987; cited in Hutson and Wagenet, 1992). The retentivity function is described by two parameters, the Campbell a and b values (constants), and the porosity (Equation 5). The a and b values are derived from the intercept and the negative slope of the retentivity curve, plotted as the natural logarithm of the matric potential against moisture content. LEACHM provides a choice of water retention regression models for predicting water retention parameters from particle size distribution, bulk density, and organic matter content. The basic Campbell's water retention equation (1974) is

$$h = a\left(\frac{\theta}{\theta_s}\right)^{-b} \qquad (16)$$

where θ_s is volumetric water content at saturation ($m^3.m^{-3}$.) and a and b are constants. In LEACHM, modifications of Equation 5 are used where the exponential function is replaced by a parabolic function to represent water retention in field soils at high matric potentials.

LEACHW takes vegetation and plant growth into account, but it is not intended to simulate crop growth. The crop cover fraction is used to partition potential evapotranspiration into potential evaporation and potential transpiration. The distribution of roots with depth partly determines the water and chemical uptake terms within each segment. The crop growth routines included in the programme are simple representations of root distribution as a function of time and depth. They are based upon empirical equations and no feedbacks occur between soil conditions and plant growth. The evapotranspiration subroutines are based upon the methods of Childs and Hanks (1975). From weekly A-pan evaporation totals and crop or pan factors, the potential surface evaporation and potential daily transpiration values are calculated. The rate of water absorption by plant roots is also simulated by LEACHM following Nimah

and Hanks (1973; cited in Hutson and Wagenet, 1992). In this calculation, transpiration, effective root water potential and root water flow resistance are accounted for.

The output from LEACHM is presented in the form of tables in .OUT files, and includes:

1) a table of profile water retentivity and hydraulic conductivity data;
2) a cumulative mass balance summary for the whole profile;
3) soil profile water contents, potentials and changes; and
4) evapotranspiration details.

In addition, output from the model can be retrieved from a summary file (.SUM) in a format suitable for use in time series plots. Output data from this file was used to produce the output graphs presented in this study.

4.2.1 Mass balance errors

LEACHM makes use of the finite difference approach to solve the Richards equation of moisture movement. This means the equation is solved, through an iterative process, for each node corresponding to each segment. At each stage of iteration it is possible that some water is lost from the balance through some mathematical artefact inherent in the process of solving the Richards equation. This loss, or error, becomes more pronounced when the simulation process is continued for an extended period. This error in the flow routine will subsequently be transferred to other subroutines in which the soil moisture status, calculated as a function of the Richards equation, is incorporated. The mass balance error also becomes more pronounced when the user specifies too few segments for the profile. A consequence of too few segments in the profile is a loss of accuracy in solving the Richards equation.

4.2.2 Some limitations of LEACHM

According to Hutson and Wagenet (1992) LEACHM is not intended to:

1) use unequal depth increments;

2) predict runoff water quality and quantity;
3) simulate the response of plants to soil or environmental change, or predict crop yields;
4) simulate the transport of immiscible liquids; or
5) predict solute distribution in situations subject to two- or three dimensional flux patterns.

LEACHM is a one-dimensional flow model and as such it does not take into account lateral water movement within the profile. In the case of Coastal Park, however, this limitation can be ignored. The proposed soil cover of Cape Flats sand is an artificial, uniform, sandy soil profile, devoid of any impermeable layers, such as clay lenses, which could cause lateral flow to take place.

LEACHM does not allow the user to specify more than one node per segment, which can mean repetitiveness when defining soil segment parameters. In the case of a uniform sandy soil profile, such as that found at Coastal Park, specifying more than one node per segment would have increased the accuracy of the output and saved time in preparing the input file, since the number of segments requiring equivalent soil parameters could have been reduced.

The user should also be aware of some of the assumptions made in preparing the input files for LEACHW. One of these is that rain falls over a 24 hour period with an intensity equal to the amount fallen, divided by the 24 hour period. This implies that sufficient time is available for all the rain to infiltrate the profile. This may not reflect the real situation and could lead to underestimation of leachate discharge. Shorter duration and higher intensity rainfall events could occur, possibly causing runoff and conditions for ponded infiltration. If ponded infiltration does occur, then water could be forced through the profile under a positive hydraulic head, causing it to bypass the plant root system, thereby reducing interception and return of moisture to the atmosphere by transpiration.

4.3 Modelling approach

LEACHW (water regime submodel of LEACHM Version 3) was employed to simulate the water balance of a hypothetical soil cover at the Coastal Park landfill. The landfill is classified as a G:L:B$^+$ waste site, i.e. it receives General municipal waste, is Large (maximum rate of deposition >500 tonnes per day), and has a positive water Balance (i.e. produces significant quantities of leachate; Novella and Eichstädt, 1995). The G:L:B$^+$ Classification requires Coastal Park to adhere to the minimum requirements for rehabilitation, closure and end-use (DWAF, 1994). This requires the design, construction and maintenance of a cover or capping system. The cover must maintain its integrity at all time and must be capable of supporting a sustainable vegetation cover. The capping system design recommended by DWAF (1994) for a G:L:B$^+$ landfill is shown in Figure 4.1 (DWAF, 1994).

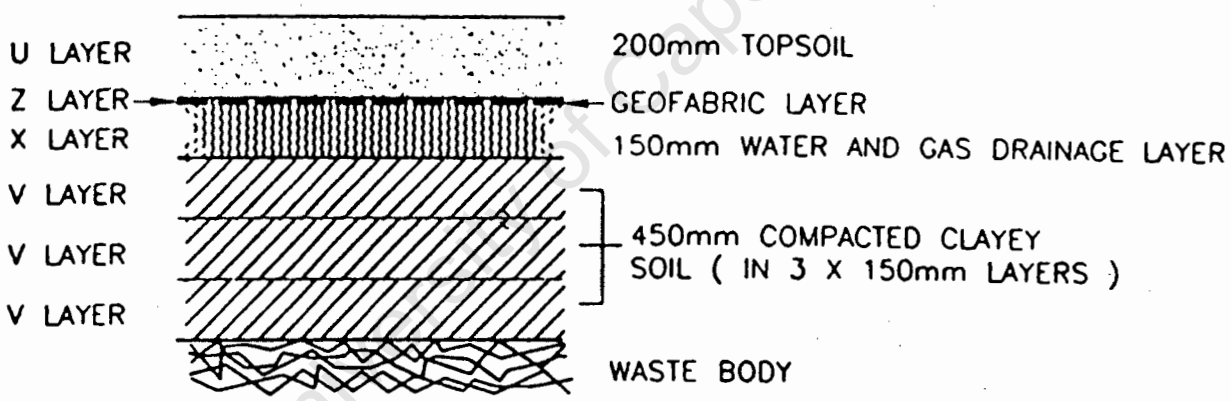

Figure 4.1 Recommended capping design specified for G:L:B$^+$, H:h and H:H landfills, according to the Minimum Requirements legislation (DWAF, 1994).

Additional cost could be incurred because of the demand for a suitable clay material for the 450 mm compacted clay layer (Fig. 4.1). It is proposed that the same effect may be obtained using less clay but more topsoil (1.0 - 2.0 m deep) for Coastal Park. A layer of

sand originating from the Coastal Park surroundings, deep enough and covered with the correct vegetation to maintain a deficit water balance, is proposed. LEACHW will be used to model only the topsoil layer.

The underlying assumptions in this study are:

1) Any water, derived from precipitation or irrigation, which percolates below the rooting zone in the cover, is considered to be drainage into the waste body and will contribute directly to leachate generation at the base of the waste body.
2) The waste body is hydraulically continuous, exists at its drained upper limit and behaves as a cascading system, i.e. tipping bucket effect.
3) There is sufficient Cape Flats Sand to provide a 70 hectare area with a soil cover depth of 2.0 m if necessary, ie. 140 000 m^3 of sand. Figure 4.2 illustrates the proposed capping system:

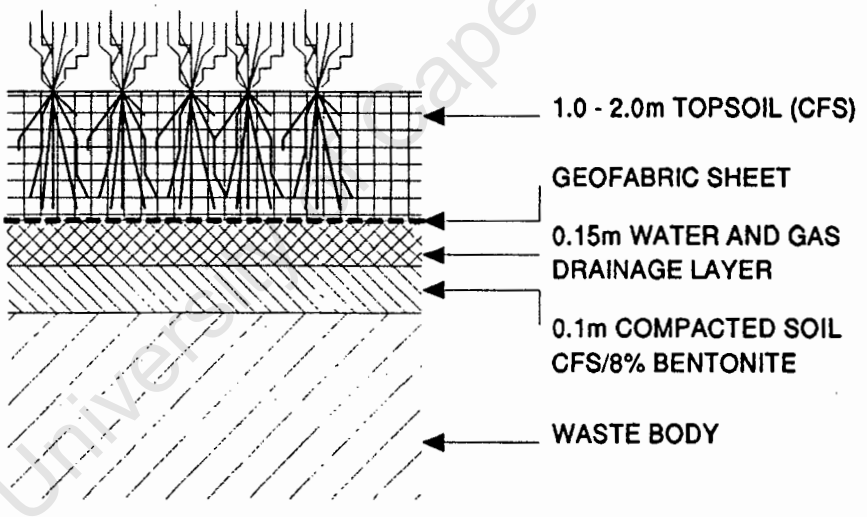

Figure 4.2 Proposed design of capping system using Cape Flats sand and a single clay treated layer. The topsoil cover has been modelled using cover depths of 1.0 and 2.0 m.

4.4 Preparation of LEACHW input file

Appendix IV contains two sample input files, corresponding to the two simulation sets conducted. The first simulation set used rainfall for the wettest year on record (1954) while the second used rainfall from the average year on record (1970). The sample file,

WATFTEST, was used as the template file and edited in the MS Dos ASCII Editor to form the input files for each simulation run. The sample input files in Appendix IV, for the 1.0 m soil depth and 90 % vegetation cover simulations, contain the soil, vegetation, evaporation and rainfall input data.

4.4.1 Soil

The input file for each simulation set was prepared assuming soil cover depths of 1.0, and 2.0 m. For each of these cover depths, percentage vegetation densities of 0, 50, 70 and 90 percent were applied. Thus, 16 simulation runs were conducted, 8 for the 1954 rainfall data and 8 for the 1970 rainfall data.

A simplified approach was to assume that the Cape Flats sand, characterised in Chapter 2, exhibits uniform hydrological characteristics throughout the soil cover depth. For instance, in the input file, a soil depth of 1.0 m was specified to have ten 10 cm segments with each segment possessing the same hydraulic conductivity, bulk density, particle size distribution and organic matter content. The retention model developed by Hutson (1986) (in Hutson and Wagenet, 1992) was chosen for predicting water retention parameters from particle size distribution, bulk density and organic matter content. Table 4.1 presents the soil parameters used in each of the simulation runs.

Table 4.1 The more important values of the soil physical parameters used in LEACHW.

Parameter	Value
Particle size distribution:	
% Clay	0.05
% Silt	0.05
% Organic matter	0.006
Starting matric potential (kPa)	-1500
Particle density ($kg.dm^{-3}$)	
Clay	2.65
Silt and sand	2.67
Organic matter	1.10
Bulk density ($kg.dm^{-3}$)	1.70
Hydraulic conductivity ($mm.day^{-1}$)	2590

4.4.2 Vegetation

The Rooikrantz (*Acacia cyclops*) is a common alien to the Cape Peninsula. It is capable of growing in very sandy soils and can tolerate very high soil matric potentials before it experiences permanent wilting. Because of its regular occurrence along the False Bay coastline, Rooikrantz was assumed in the model to be the plant growing on the soil cover. The plant's tolerance to dry conditions, its high transpiration rate and its evergreen foliage make it potentially important in maintaining a deficit water balance of the landfill capping system. The system has been modelled using 0, 50, 70 and 90 percentage Rooikrantz cover. A 0 percent vegetation cover was set to assess what the probable leachate generation would be during and after a rainfall period following a severe fire. *Acacia cyclops* relies on combustion by fire for its propagation and germination, and is quick to invade open areas recently cleared by fire, rapidly establishing rooting systems and canopy cover. Thus it could be expected that significant leachate generation following a fire would be short term, lasting only until the vegetation has established mature rooting systems. Simulations have been run assuming mature vegetation cover with a static, well-established rooting system and crop cover.

4.4.3 Rainfall

Annual rainfall data was chosen to represent the wettest and average year winter rainfall seasons (May to October). The *Minimum Requirements* (DWAF, 1994) recommend the use of such data as a conservative approach for water balance studies at landfill sites. Rainfall data was extracted from the CCWR rainfall database for the Rondevlei rainfall station (34°04 S, 18°30 E), which has a mean annual precipitation (MAP) of 664.6 mm and is the closest meteorological station to Coastal Park with a rainfall record length of 42 years. From the unpatched data in the record, the year 1954 had the highest rainfall of 1065.1 mm, and 1970 was chosen, subjectively, as having close to an average rainfall of 644.3 mm.

Version 3 of LEACHM has a dimension statement imbedded in its Fortran source code which limits the specified number of rainfall/irrigation events. In this case the model is limited to the use of 60 rainfall events. This prohibited the use of a continuous rainfall

record extending throughout an entire year or an entire season. As a result, only the central 60 rainfall events of the winter season were chosen as input for both 1954 and 1970. Due to time constraints and the unavailability of direct expertise, the code was not adjusted to allow for a more extended rainfall period. The daily rainfall (mm) periods selected from years 1954 and 1970 for simulations using LEACHW are indicated in Figures 4.3 and 4.4.

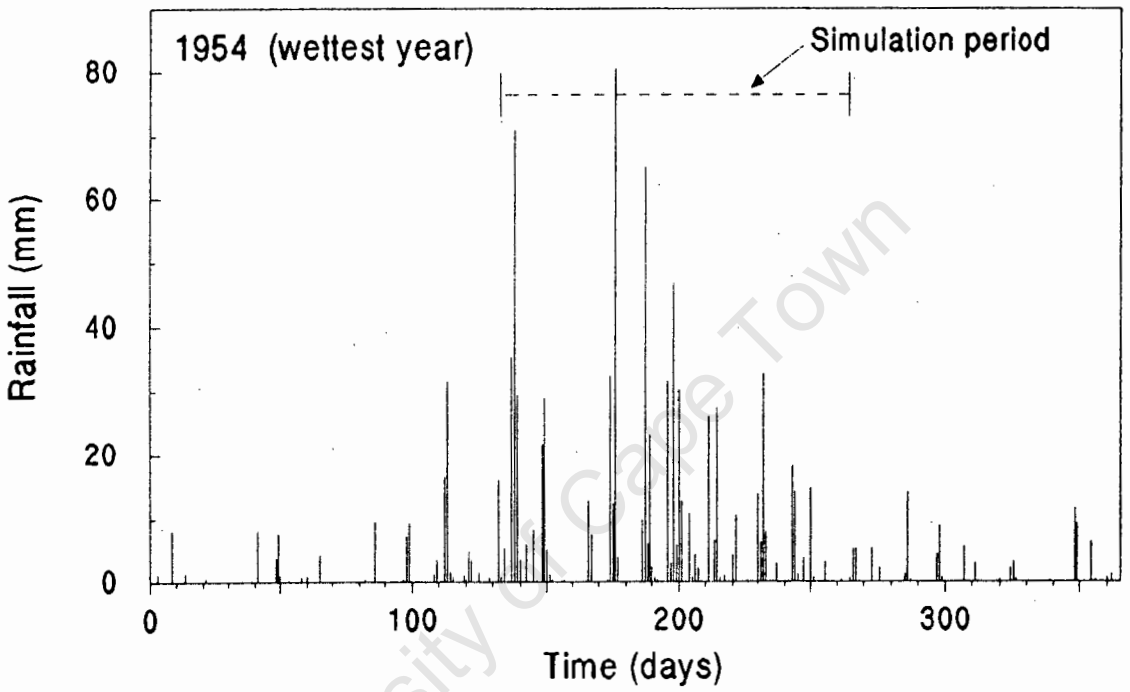

Figure 4.3 Observed daily rainfall (mm) recorded at Rondevlei (station ID 004874W) in 1954, showing the 127-day, 60-rainday period selected for simulation using LEACHW.

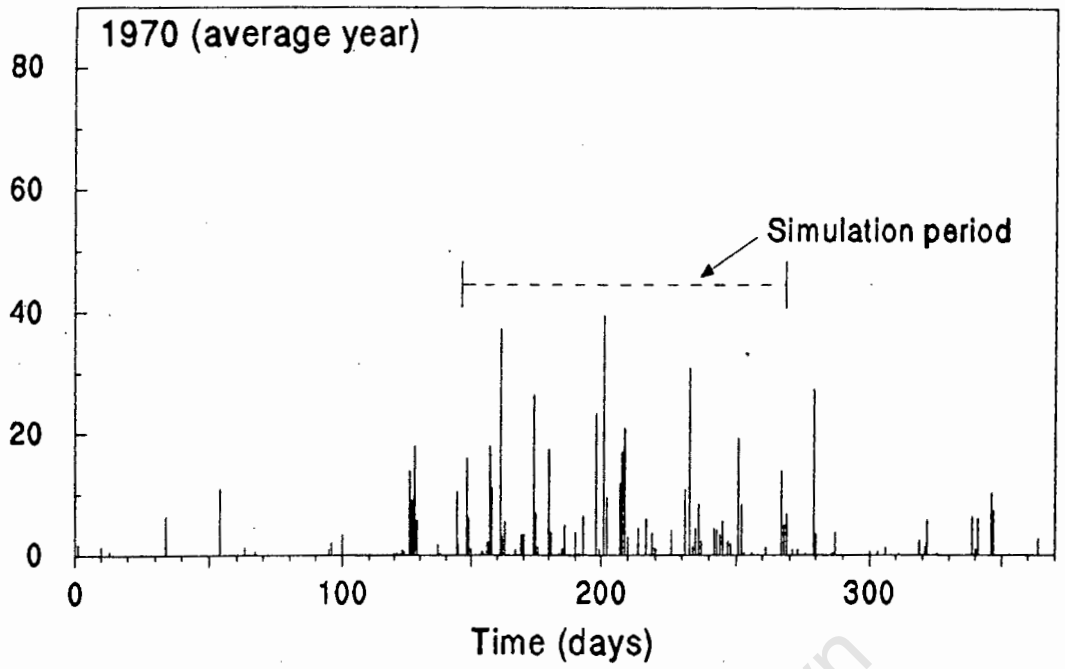

Figure 4.4 Observed daily rainfall (mm) recorded at Rondevlei (station ID 004874W) in 1970, showing the 122-day, 60-rainday period selected for simulation using LEACHW.

4.4.4 Evaporation

Daily A-pan evaporation was also extracted from the CCWR daily evaporation and temperature database. Rondevlei did not have any recorded A-pan data and the nearest meteorological station, Station A5032 (34°02 S, 18°32 E) was considered. Daily evaporation A-pan data was only available for the year 1954 and was used in both 1954 and 1970 simulation runs. The data were imported into a spreadsheet where they were converted into weekly totals as required for input into LEACHW.

4.5 Results and discussion

LEACHW was used to simulate the water balance of the proposed soil cover under two rainfall conditions, ie. the wettest year on record (1954) and an approximately average rainfall year (1970). Sample output files of the LEACHW simulations are presented in

Appendix V for both the 1954 and 1970 wet seasons with a 90% vegetation cover established on a soil cover of 1.0 m depth. The output file is generated after specified time intervals during modelling. Tables 4.2 and 4.3 show the cumulative leachate generated after 60 rainfall events within the winter rainfall seasons (May to October) of 1954 and 1970, respectively.

Table 4.2 Simulated cumulative leachate (in mm) generated by the varying soil cover depths (m) and vegetation covers (%) at Coastal Park after the *wettest* year's rainfall season (1954, May to October).

Soil Cover	Vegetation cover (%)			
depth (m)	90	70	50	0
1.0	339	345	365	437
2.0	200	204	218	278

Table 4.3 Simulated cumulative leachate (in mm) generated by the varying soil cover depths (m) and vegetation covers (%) at Coastal Park after the *average* rainfall year's wet season (1970, May to October).

Soil Cover	Vegetation cover (%)			
depth (m)	90	70	50	0
1.0	25	29	43	108
1.5	0	0	0	0

4.5.1 Effect of vegetation cover on leachate generation

Time-series plots, using data from the .SUM files, are presented in Figures 4.5 to 4.8. Predicted cumulative leachate depth (mm) as well as cumulative precipitation is plotted against time. The graphs are discussed in the light of those values presented in Tables 4.1 and 4.2.

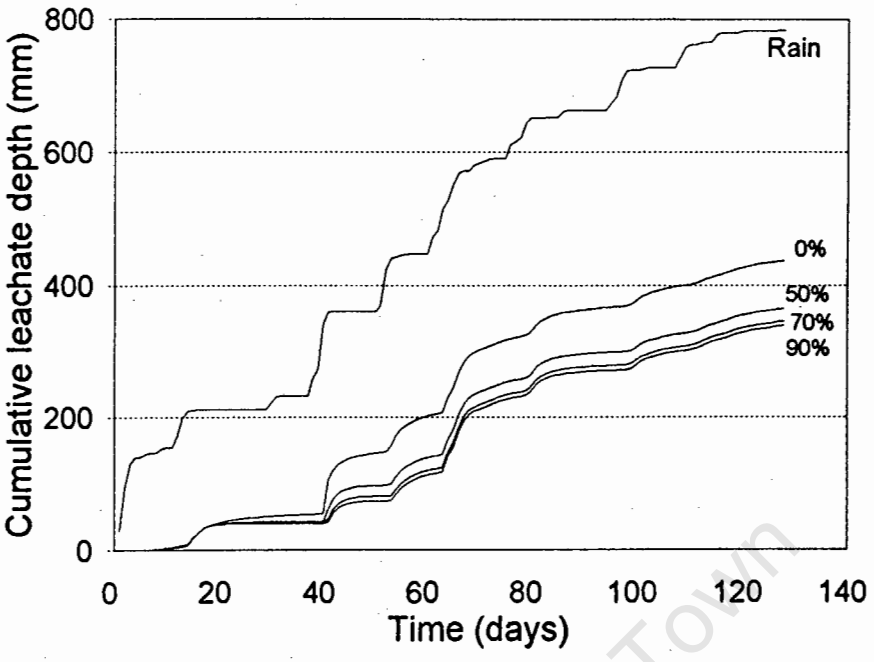

Figure 4.5 Cumulative rainfall (mm) and predicted cumulative leachate (mm) for the rainfall season of 1954 at Coastal Park, for 1m deep soil cover and various vegetation covers (%).

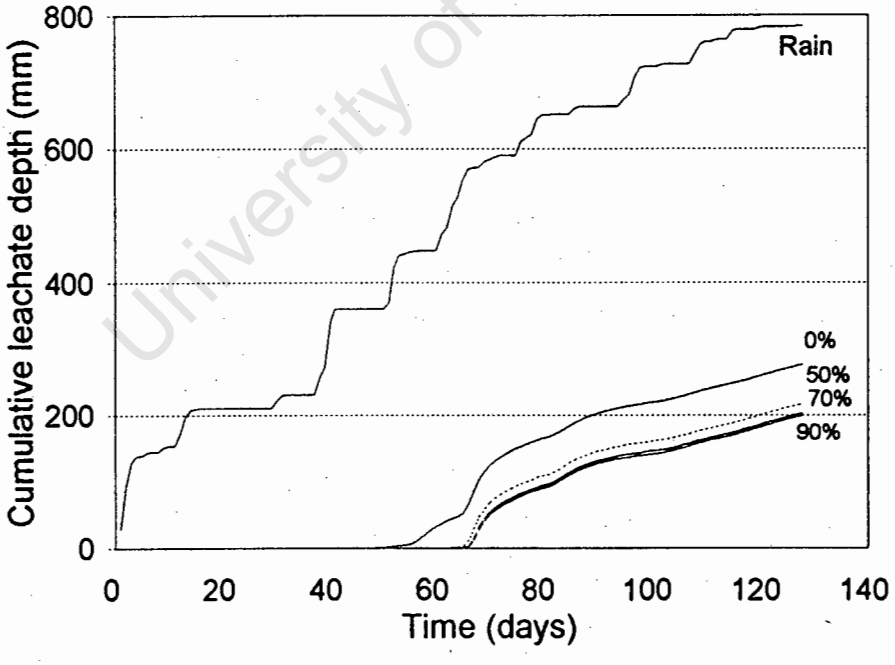

Figure 4.6 Cumulative rainfall (mm) and predicted cumulative leachate (mm) for the rainfall season of 1954 at Coastal Park, for 2m soil cover depth and various vegetation covers (%).

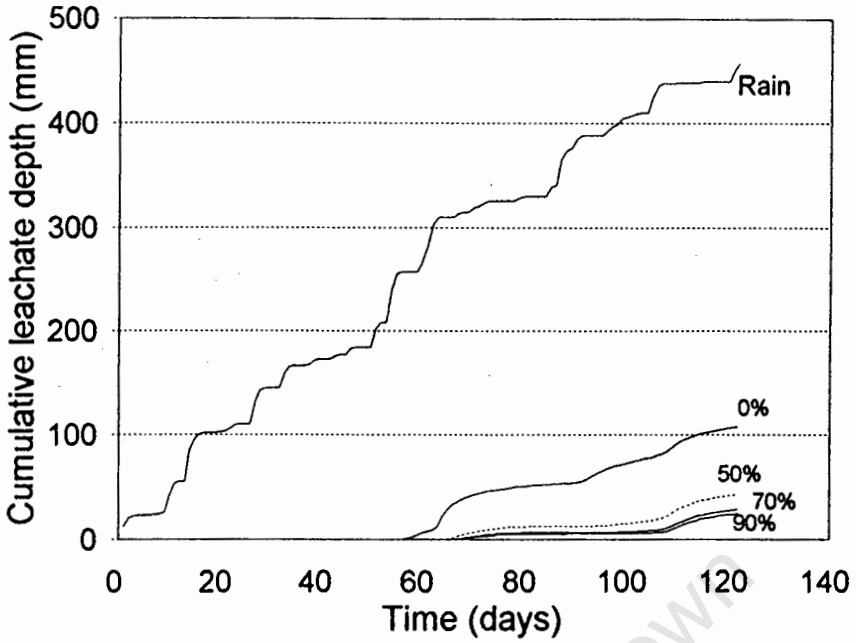

Figure 4.7 Cumulative rainfall (mm) and predicted cumulative leachate (mm) for the rainfall season of 1970 at Coastal Park, for 1m soil cover depth and various vegetation covers (%).

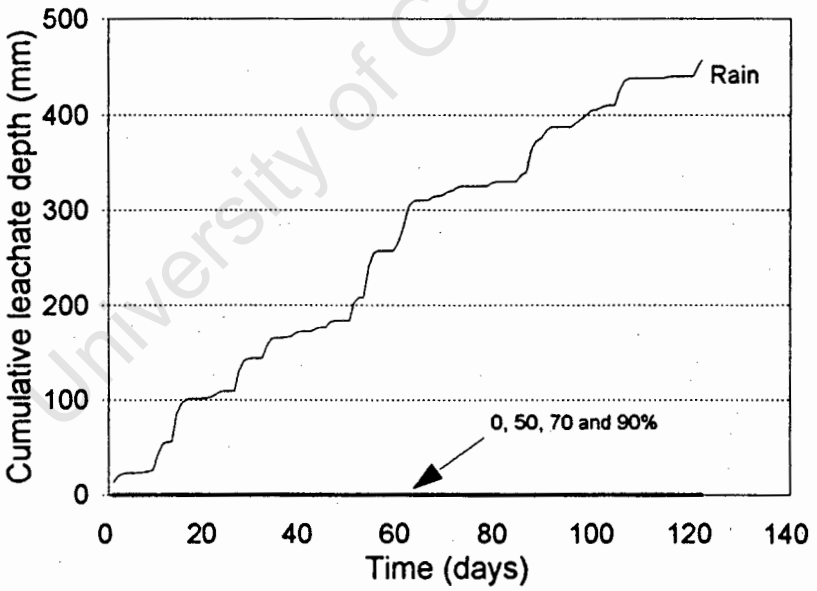

Figure 4.8 Cumulative rainfall (mm) and predicted cumulative leachate (mm) for the rainfall season of 1970 at Coastal Park, for 2m soil cover depth and various vegetation covers (%). The solid line corresponds to zero predicted values.

Leachate generation, predicted from the 60 rainfall events occurring within the rainfall season of 1954 over a 127 day period, is graphically presented in Figure 4.5 and 4.6. Figure 4.5 shows the effect that a 1.0 m soil cover depth and varying percent vegetation cover has on the quantity of leachate generated. Where a 1.0 m soil cover is used, LEACHW predicts that between 338 and 437 mm of leachate will be generated when the vegetation cover ranges between 90 and 0 %, respectively. Increasing vegetation cover from 0 to 50 % results in a 16 % reduction in leachate discharge. Improving vegetation cover from 50 to 90 % will effectively reduce leachate generation by only a further 6%. Under average rainfall conditions (1970), LEACHW predicts that improving the vegetation cover on the landfill from 0 to 90 % will reduce leachate discharge from 108 to 25 mm (Fig. 4.7). This represents a reduction of almost 77 %. 60 % of this reduction is achieved with a 50 % vegetation cover, whereas an improvement from 50 to 90 % vegetation cover is only predicted to reduce the quantity of leachate generated by a further 16 %.

Under the wettest conditions, a 2.0 m soil cover (Fig. 4.6) is still not sufficient to prevent saturated conditions from developing, and loss of water from the profile subsequently contributes to leachate generation. A 90% vegetation cover is predicted to reduce leachate generation from 278 (0% vegetation cover) to 200 mm. This 28% reduction using 2.0 m of topsoil is only a slight improvement on the 22% reduction in leachate under the same conditions but with 1.0 m topsoil cover. Again, LEACHW predicts that the greatest percent reduction (about 22%) is achieved by improving vegetation cover from 0 to 50%. Leachate is only reduced by a further 8% when vegetation cover is improved from 50 to 90%.

An average rainfall season such as the 1970 season, represents far more favourable conditions (Fig. 4.8). In this case, the model predicts that a 2 m soil cover depth, even with a 0% vegetation cover, will be characterised by a deficit water balance, in which no drainage from the profile will occur and hence no leachate will be generated.

Figure 4.9 summarises the predicted cumulative leachate after 60 rainfall events, showing the different extents to which vegetation cover is effective in reducing leachate discharge under average and very wet conditions, and at 1 and 2 m soil cover depths.

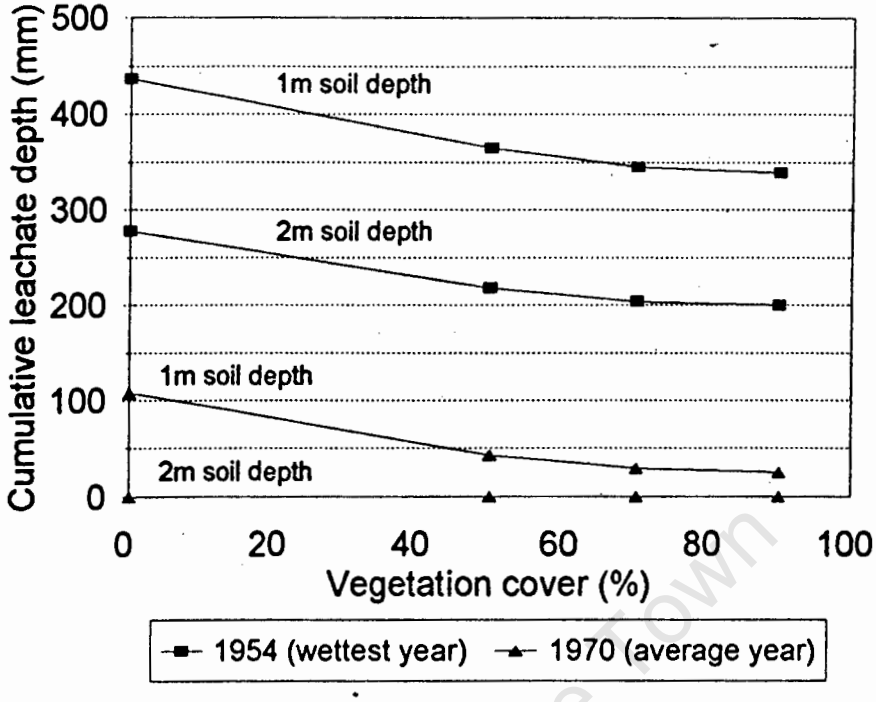

Figure 4.9 Relationship between predicted leachate depth (mm) and vegetation cover (%) with 1 and 2m soil cover depths for the average (1970) and wettest (1954) years.

4.9 Conclusions

LEACHW has been used in a preliminary water balance modelling exercise which has revealed some noteworthy relationships between quantities of leachate generated, topsoil depth and vegetation cover. The model has shown that variation in topsoil depth and vegetation cover can reduce or enhance the leachate producing capacity of a landfill by controlling the amount of moisture entering the waste pile. During the wettest season a combination of 90% vegetation and 2 m topsoil will only reduce the quantity of leachate generated by 28% relative to no vegetation cover. Under average rainfall conditions, a 77% reduction in leachate generation can be achieved with a 90% vegetation cover and 1 m of topsoil, relative to no vegetation cover. No leachate will be generated with a 2 m topsoil under the same conditions, even with no vegetation.

As far as groundwater pollution by leachate is concerned, the results of this exercise have shown that under average rainfall conditions, it could be feasible to use a 1 to 2 m capping system with a good vegetation cover as a means of significantly reducing leachate generation from the landfill. However, under extreme rainfall conditions, it is almost certain that such a system would have to be augmented by additional leachate management strategies, such as an effective leachate drainage and collection system, to eliminate or reduce leachate generation from the landfill and subsequent groundwater contamination.

The model computed actual moisture movement under saturated and unsaturated conditions, using refined equations such as the Richards equation. It can therefore be considered an improved approach on the common water balance methods often used at landfills for leachate prediction, which do not consider any moisture flow. It should be remembered that the moisture dynamics of a proposed capping system have been modelled here, and not the entire landfill and waste body. Furthermore, the model has not been calibrated against real-time leachate measurements. For the purposes of model calibration on a field scale, use should be made of landfill experimental cells (similar to those described by Blight, 1995), where varying capping systems and degrees of vegetation cover can be constructed to specification and measurements of leachate flow made and compared with LEACHW predicted values.

Finally, it should be remembered that this has been a modelling exercise in which predictions are made under a set of hypothetical conditions, and thus does not necessarily reflect the real situation. However, it has pointed out some interesting possibilities worth exploring further which could become cost effective when the design and construction of a landfill capping system under similar conditions is considered. For instance, a slightly thinner clay liner or clay-treated layer (0.1 m 8 % bentonite/sand layer as in Fig. 4.2) than that recommended by the Department of Water Affairs and Forestry could be implemented where the abundance and accessability of soil in the area can allow a deeper topsoil to be constructed.

Chapter 5

Conclusions and recommendations

Soil at the Coastal Park landfill site can be classified as an aeolian, calcareous medium sand with negligible organic matter and poor structure. Although good pH-buffering is imparted through an abundance of calcium carbonates, the capacity for the soil to attenuate leachate can be expected to be minimal because of a negligibe content of clay-size particles.

Because of its similarity in colour (dark green) and its instability in air, it has been concluded that the leachate contains compounds consisting of Fe^{2+} - Fe^{3+} hydroxides known as green rusts. It is suggested that the characteristic darkness of the leachate is also attributable to amorphous sulfides of iron (such as mackinawite and greigite) and other heavy metals, as well as organic compounds. The sulfides are easily oxidized upon exposure to air, causing the isolated leachate solid phase to undergo a colour change from black to orange-brown as iron oxides form. Centrifugation and freeze-drying is an acceptable, non-oxidative technique for isolation and chemical characterisation, including analysis by X-ray diffractometry, of the leachate solid phase. The leachate was found to contain 400 mg solids per litre. Different methods of solid phase preparation for XRD analysis can result in different diffraction patterns. Calcite was identified as one of the main constituents when a smear of the solid phase "paste" was prepared on a glass slide for XRD analysis, whereas halite was the main constituent identified from analysis of a freeze-dried powder-mount. Sodium, calcium and chloride are present in high concentrations in the leachate and it was concluded that CO_2 dissolved in the paste as it dried, reacting with Ca in the leachate to produce calcite.

Hydraulic conductivity of the soil, estimated on a laboratory-scale, was an order of magnitude higher than that determined geohydrologically by previous workers in the field. The hydraulic conductivity (K) of untreated soil can be reduced by almost three orders of magnitude by leaching the soil with landfill leachate, a minimum value of $10^{-4.5}$ cm.s^{-1} having been achieved. This reduction is in response to pore blockage by suspended leachate solids. The hydraulic conductivity of the soil both before and after treatment

with leachate, however, is such that a need exists for some form of soil amendment before further waste tipping can take place.

Of the various soil amendments tested, an 8 % kaolinite plus 4 % gypsum treatment was the most effective, maintaining a minimum K of $10^{-4.5}$ cm.s^{-1}, which, however, is still higher than the local requirement of 1 x 10^{-7} cm.s^{-1}. Amendment with 8 % Na-bentonite initially achieved a minimum K of $10^{-7.8}$ cm.s^{-1}, but the high electrical conductivity (EC) of the leachate (26.8 mS.cm^{-1}) caused shrinking and severe side-wall seepage, which rapidly enhanced hydraulic conductivity, reaching a maximum K of about $10^{-4.7}$ cm.s^{-1}. Both treatments of the sand do show promise as possible liners, although the use of higher percentage concentrations of clay should be investigated further. Where no gypsum was added to the kaolinite-treated soil, results showed that too much dispersion and purging of clay from the column in response to Na_2CO_3 solution can destroy any effects of sealing by the kaolinite particles. The bentonite-treated soil could reflect a real situation in which a bentonite liner might undergo salinity-induced shrinking and cracking in response to the high EC of the leachate, followed by enhanced hydraulic conductivity and loss in integrity as a liner.

The extent to which a suitable landfill capping system and vegetation cover would influence leachate generation at the Coastal Park landfill has been tested using LEACHW (water regime submodel of LEACHM). The aim was to design the soil capping system so as to achieve a negative water balance which would ideally prevent the ingression of rainwater into the waste body, thereby inhibiting leachate generation. The model predicted that under average rainfall and evaporation conditions the landfill, with a 2 m soil depth and 0 % vegetation cover, would still not generate leachate. However, under the wettest conditions not even a 90% vegetation cover and 2 m soil cover would be sufficient to prevent the landfill from generating leachate, suggesting that, under such conditions, a more effective leachate management strategy, such as leachate collection sumps, should be implemented. This exercise demonstrated that the application of LEACHM to vegetated soil capping systems is an alternative means of predicting leachate discharge from landfill sites.

Generally, it can be concluded that the soil found in the region of Coastal Park is incapable of any significant attenuation of landfill leachate. The high hydraulic conductivity of the sandy soil facilitates the efficient migration of pollutants to the groundwater, a known problem at Coastal Park. Even if some of the suspended solids associated with leachate are filtered out by the fine sand fraction, leachate colloid particles are still capable of passing through the porous medium to the groundwater, carrying with them adsorbed heavy metals, such as Cu, Cr, Ni, and Pb. It is highly recommended that at least some form of liner system should be constructed before any further extensions of the tipping area are made. Such a liner would help reduce the problem of leachate migration and subsequent groundwater pollution, which may conceivably continue well into the next century.

Finally, there are possible areas of future work which could be recommended as a follow up to this study:

1) The effect of increased kaolinite content on the hydraulic conductivity of the Coastal Park sand warrants further investigation. Twelve and possibly 16 % kaolinite/sand mixtures could be used, with and without a 4 % gypsum-treated middle layer, and using fresh water to disperse the clay as opposed to the Na_2CO_3 used in this study. This would establish whether prior soil treatment with a dispersant is necessary for seal formation before leaching with landfill leachate.

2) The modelling exercise conducted in this study has been of a preliminary nature and room for further refinement exists. Firstly, the model should be run using an adjusted source code which will allow the input of a continuous rainfall record extending beyond 60 rainfall events; at least a year's rainfall record should be accommodated. Secondly, the model should be calibrated against observed leachate flow data. For this purpose, a series of experimental landfill cells could be constructed, similar to those described by Blight (1995), but with vegetation covers and soil capping systems similar to those proposed in this study. Leachate discharge from these cells can be measured and compared to values predicted by LEACHW.

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Appendix I

INSTRUMENTAL PARAMETERS AND DATA QUALITY FOR ROUTINE MAJOR AND TRACE ELEMENT DETERMINATIONS ON SILICATES BY WDXRFS

by

J P WILLIS

MAJOR ELEMENTS

Nine major elements, Fe, Mn, Ti, Ca, K, P, Si, Al and Mg (with Ni and Cr when Ni and Cr concentrations exceed ~2000 ppm) are determined using fusion disks prepared according to the method of Norrish and Hutton (1969). The disks are analyzed on a Philips PW1480 wavelength dispersive XRF spectrometer with a Mo/Sc x-ray tube. Fe, Mn and Ti are measured with the tube at 100 kV, 25 mA. The other elements are determined with the tube at 40 kV, 65 mA. Peak only measurements are made on the elements Fe through Mg. Sodium is determined using powder briquettes, the x-ray tube at 40 kV, 65 mA, and with backgrounds measured at −2.00 and +2.00°2θ from the peak position. Analytical conditions are given in Table 1.

Fusion disks made up with 100% Johnson Matthey Specpure SiO_2 are used as blanks for all elements except Si. Fusion disks made up from mixtures of Johnson Matthey Specpure Fe_2O_3 and $CaCO_3$ are used as blanks for Si. Intensity data are collected using the Philips X40 software. Matrix corrections are made on the elements Fe through Mg using the de Jongh model in the X40 software. Theoretical alpha coefficients used in the de Jongh model for all other elements on the analyte element are calculated using the Philips on-line ALPHAS programme. Na_2O is not included in the matrix corrections in de Jongh model, and no matrix corrections are made to the sodium intensities.

Table 1. Analytical conditions for determination of major elements using a Philips PW1480 WDXRF spectrometer.

Element /line	Collimator	Crystal	Detector	PHS LWL	PHS UPL	Counting time (s)	Concentration range *	RMS	No. of standards
FeKα	F	LiF(220)	FL	16	70	150	0 - 17	0.118	14
'MnKα	F	LiF(220)	FL	15	70	150	0 - 0.22	0.005	14
TiKα	F	LiF(200)	FL	28	70	150	0 - 2.75	0.020	14
CaKα	F	LiF(200)	FL	36	70	20	0 - 12.5	0.037	14
KKα	F	LiF(200)	FL	36	70	50	0 - 15.5	0.057	14
PKα	C	GE(111)	FL	25	75	100	0 - 0.36	0.008	14
SiKα	C	PE(002)	FL	32	74	100	0 - 100	0.408	14
AlKα	C	PE(002)	FL	25	75	80	0 - 17.5	0.136	14
MgKα	F	PX-1	FL	30	74	150	0 - 46	0.095	14
NaKα	F	PX-1	FL	30	78	200	0 - 9	0.189	15

* = all concentrations expressed as wt% oxide

$$RMS = \sqrt{\frac{1}{n - k} \sum (Conc_{given} - Conc_{calc})}$$

where

n	=	no. of standards
k	=	no. of calibration coefficients, *i.e.* 2, the slope and intercept of the calibration line.
$Conc_{given}$	=	recommended concentration for an element in a standard
$Conc_{calc}$	=	concentration of an element calculated from the best-fit calibration line

First order calibration lines, with intercept, are calculated using all data points, including blanks. Calibration plots for Fe_2O_3, CaO, SiO_2 and MgO are given in Figures 1 - 4.

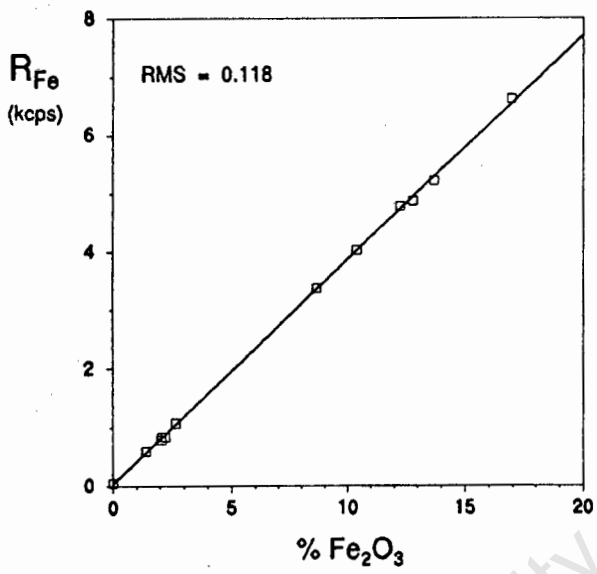

Figure 1. Calibration plot for Fe_2O_3 using "Norrish" fusion disks.

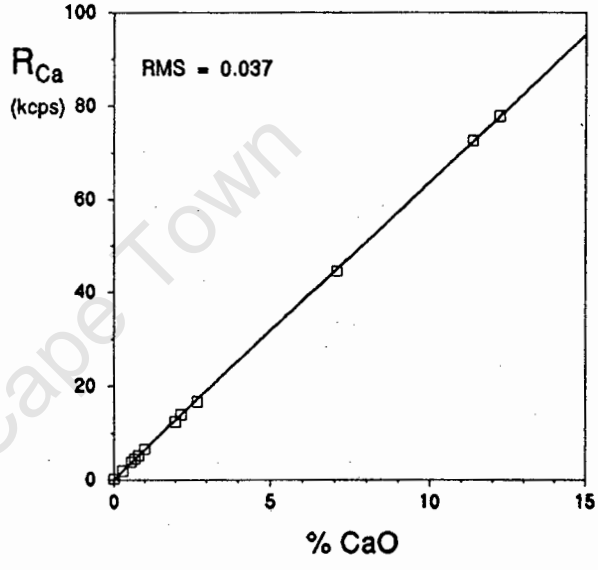

Figure 2. Calibration plot for CaO using "Norrish" fusion disks.

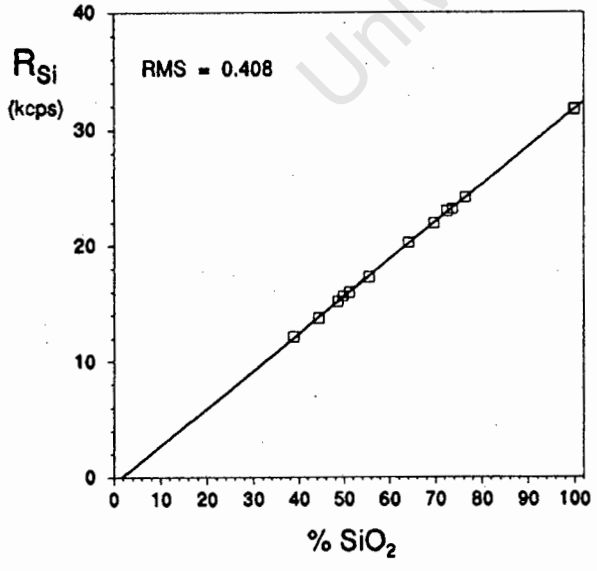

Figure 3. Calibration plot for SiO_2 using "Norrish" fusion disks.

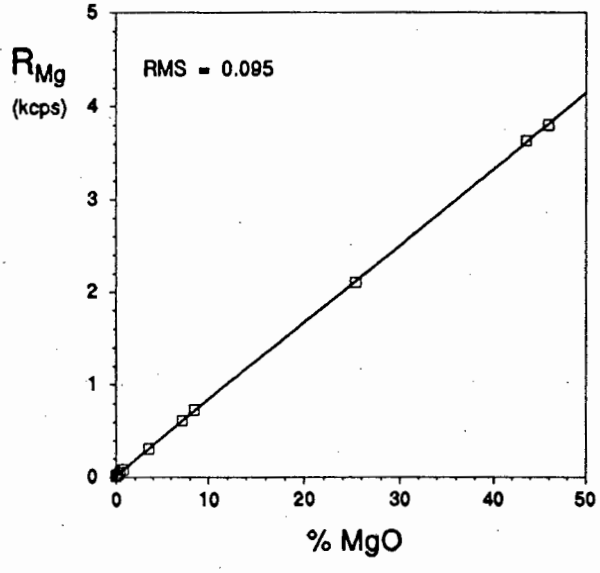

Figure 4. Calibration plot for MgO using "Norrish" fusion disks.

TRACE ELEMENTS

Trace elements are determined on powder briquettes using a series of x-ray tubes. Analytical conditions are listed in Tables 2 and 3.

Table 2. X-ray tubes and tube and x-ray path settings for the determination of trace elements using a Siemens SRS303AS and Philips PW1480 WDXRF spectrometer.

Spectrometer	Element/line	X-ray tube		X-ray path
		Target	kV - mA	
SRS303AS	RhKαC	Rh	60 45	Vacuum
SRS303AS	MoKα	Rh	60 45	Vacuum
SRS303AS	NbKα	Rh	60 45	Vacuum
SRS303AS	ZrKa	Rh	60 45	Vacuum
SRS303AS	Y Kα	Rh	60 45	Vacuum
SRS303AS	SrKα	Rh	60 45	Vacuum
SRS303AS	U $L\alpha_1$	Rh	60 45	Vacuum
SRS303AS	RbKα	Rh	60 45	Vacuum
SRS303AS	Th $L\alpha_1$	Rh	60 45	Vacuum
SRS303AS	Pb $L\beta_1$	Rh	60 45	Vacuum
PW1480	ZnKα	Au	60 45	Vacuum
PW1480	CuKα	Au	60 45	Vacuum
PW1480	NiKα	Au	60 45	Vacuum
PW1480	CoKα	W	50 55	Vacuum
PW1480	MnKα	W	50 55	Vacuum
PW1480	CrKα	W	50 55	Vacuum
PW1480	V Kα	W	50 55	Vacuum
PW1480	Ba $L\alpha_1$	Cr	50 55	Vacuum
PW1480	ScKα	Cr	50 55	Vacuum

Table 3. Instrumental conditions for determination of trace elements using a Siemens SRS303AS and Philips PW1480 WDXRF spectrometer.

Element /line	Collimator	Crystal	Detector	PHS LWL	PHS UPL	Counting time (s)	Background position(s) relative to peak position	Concentration range *
RhKαC	F	LiF(220)	SC	0.6	1.5	200		
MoKα	F	LiF(200)	SC	0.7	1.7	160	−0.8 +0.65	0 - 5.2
NbKα	F	LiF(200)	SC	0.5	1.6	160		0 - 268
ZrKa	F	LiF(200)	SC	0.5	1.6	160		0 - 1210
YKα	F	LiF(200)	SC	0.5	1.6	160	−0.61 +0.54	0 - 143
SrKα	F	LiF(200)	SC	0.5	1.6	160	+0.60	0 - 440
U $L\alpha_1$	F	LiF(200)	SC	0.5	1.6	160		0 - 15
RbKα	F	LiF(200)	SC	0.5	1.6	160	+0.53	0 - 530
Th $L\alpha_1$	F	LiF(200)	SC	0.5	1.6	160		0 - 51
Pb $L\beta_1$	F	LiF(200)	SC	0.4	1.4	160	+1.27	0 - 40
ZnKα	F	LiF(220)	FS	20	80	200	−1.08 +4.24	0 - 235
CuKα	F	LiF(220)	FS	20	80	200	+4.44	0 - 227
NiKα	F	LiF(220)	FS	20	80	200	+2.52	0 - 630
CoKα	F	LiF(220)	FL	15	75	200	+1.00	0 - 116
MnKα	F	LiF(220)	FL	15	75	200	−2.30 +4.70	0 - 1700
CrKα	F	LiF(220)	FL	15	75	200	−4.10 +2.90	0 - 465
VKα	F	LiF(220)	FL	13	67	200	+3.40	0 - 640
Ba $L\alpha_1$	F	LiF(200)	FL	25	75	200	−5.20	0 - 2680
ScKα	F	LiF(200)	FL	25	75	200	−2.78	0 - 54

* = all concentrations expressed as part per million (ppm or $mg.kg^{-1}$)

The RhKα Compton peak is used to determine the mass absorption coefficients of the specimens at the RhKαC wavelength (Figure 5) and the calculated values are used to correct for absorption effects on the Mo, Nb, Zr, Y, Sr, U, Rb, Th, Pb, Zn, Cu and Ni analyte wavelengths. Primary and secondary mass absorption coefficients for the Co, Mn, Cr, V, Ba and Sc analyte wavelengths are calculated from major element compositions using the tables of Heinrich (1986). Mass absorption coefficient corrections are made to the net peak intensities, (gross peak intensities corrected for dead time losses, background and spectral overlap), to correct for absorption differences between standards and specimens. No corrections are made for enhancement, which could be small but significant (< ~5% relative) for the elements Cr, V, Ba and Sc in certain specimens, depending on their concentrations of Fe, Mn and Ti.

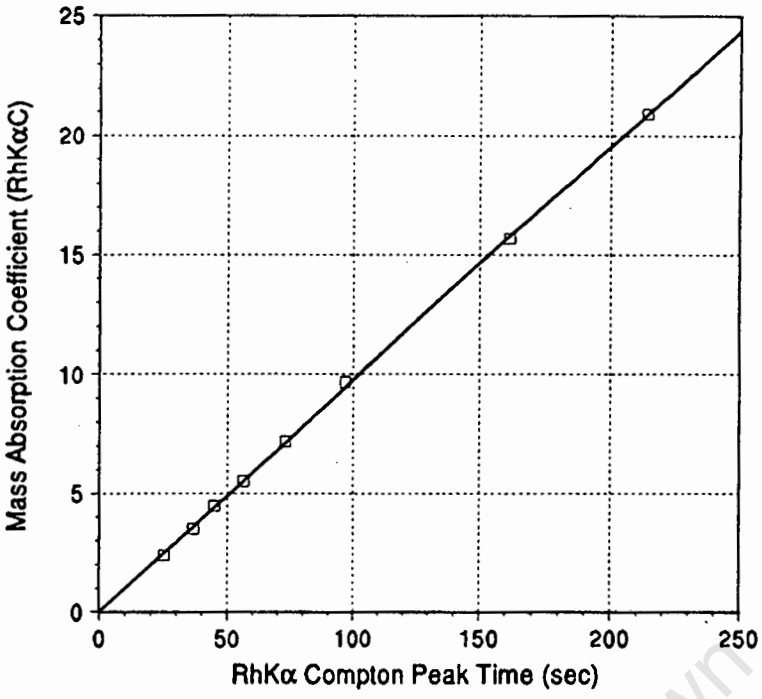

Figure 5. Calibration line for determination of mass absorption coefficients at the RhKαC wavelength. RhKαC peak time is the time required to accumulate 400 000 counts on the RhKαC peak using the fixed count method.

Measured intensity data are processed through the computer program TRACE to correct gross peak intensities for background and spectral overlap and to make mass absorption coefficient corrections according to the methods outlined in Duncan *et al.* (1984). First order calibration lines with zero intercept are calculated using six or more international rock standard reference materials (SRMs) for each element. The one standard deviation (1 σ) error due to counting statistics and the lower limit of detection is calculated for each element in each specimen.

Table 4 lists the given and calculated concentrations for selected elements in a number of rock SRMs, which gives an indication of the accuracy of the trace element data. Table 5 lists the one standard deviation counting error and lower limit of detection for each of the elements in an acidic (low Fe, Ca and Mg, high Si) rock and in a mafic (high Fe, Ca and Mg, low Si) rock. Because of the difference in mass absorption coefficients between the two types of specimen the counting error and lower limit of detection will be slightly higher in mafic rock specimens. The two examples given cover the range of mass absorption coefficients found in the majority of geological rock, soil and sediment specimens.

The counting error and lower limit of detection are calculated using the following formulae:

$$1\,\sigma\ error\ (in\ ppm) = Conc \times \frac{\sqrt{\frac{I_p}{T_p} + \frac{I_b}{T_b}}}{I_n}$$

and

$$LLD\ (in\ ppm) = \frac{6}{m}\sqrt{\frac{I_b}{T_{total}}}$$

where Conc = calculated concentration in ppm
m = net peak / concentration
I_p = gross peak count rate in cps
I_b = background count rate under the peak in cps
I_n = $I_p - I_b$ = true **net** peak count rate in cps
T_p = counting time for peak in seconds
T_b = total counting time for background in seconds
$T_{total} = T_p + T_b$

N.B. I_b is the calculated background *plus* any corrections for spectral interference, and is equal to $I_p - I_n$.

Table 4. Given and calculated trace element data (all values in ppm) for some rock SRMs.

Element	QLO-1		BHVO-1		W-2		STM-1		BIR-1	
	Given	Calc	Given	Calc	Given	Calc	Given	Calc	Given	Calc
Mo	2.6	3.5	1.0	0.8	(0.6	0.5	5.2	3.1	(0.5	<0.8
Nb	10	11	19	19	7.9	7.4	268	267	0.6	0.9
Zr	185	190	179	181	94	95	1210	1220	16	19
Y	24	25	28	28	24	23	46	47	16	17
Sr	336	329	403	395	194	195	700	689	108	109
U	1.9	2.3	0.4	<1.6	0.5	<1.2	9.1	8.8	0.01	<1.2
Rb	74	71	11	9.7	20	20	118	114	0.3	<0.6
Th	4.5	4.0	1.1	1.8	2.2	2.7	31	31	0.03	<1.5
Pb	20	20	2.6	3.1	9.3	8.5	18	17	3	3.1
Zn	61	61	105	106	77	79	235	242	71	69
Cu	29	25	136	139	103	108	(4.6	2.1	126	132
Ni	(5.8	1.8	121	127	70	72	(3	1.7	166	170
Co	7.2	7.6	45	44	44	43	0.9	<1.9	51	52
Mn	720	690	1300	1290	1260	1240	1700	1600	1320	1280
Cr	(3.2	3.6	289	312	93	100	(4.3	3.2	382	404
V	54	44	317	314	262	257	(8.7	<1.6	313	306
Ba	1370	1430	139	138	182	191	560	589	7.0	10
Sc	8.9	10.3	31.8	33.9	35	36	0.6	0.5	44	39

Table 5. **Calculated trace element data, 1σ counting error and lower limit of detection (all values in ppm) for two rock specimens having different mass absorption coefficients.**

Element	JR-2			JB-1a		
	Calc	1 σ	LLD	Calc	1 σ	LLD
Mo	4.1	0.2	0.6	1.8	0.3	0.7
Nb	19	0.1	0.4	28	0.2	0.5
Zr	87	0.1	0.3	152	0.2	0.4
Y	51	0.2	0.6	24	0.2	0.6
Sr	8.2	0.1	0.4	444	0.3	0.5
U	11	0.3	0.9	2.3	0.4	1.2
Rb	303	0.2	0.4	39	0.2	0.6
Th	34	0.4	1.1	9.8	0.5	1.4
Pb	24	0.5	1.3	7.5	0.6	1.8
Zn	28	0.2	0.6	84	0.4	0.9
Cu	1.1	0.3	0.8	55	0.5	1.1
Ni	1.3	0.3	0.8	139	0.7	1.3
Co	<1.2	0.4	1.2	37	0.9	2.3
Mn	878	1.7	1.2	1100	2.0	1.8
Cr	1.6	0.4	1.3	406	1.5	2.0
V	1.7	0.4	1.2	193	1.4	3.0
Ba	28	0.6	1.5	523	1.8	3.3
Sc	6.0	0.2	0.5	26	0.4	0.9

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Appendix II

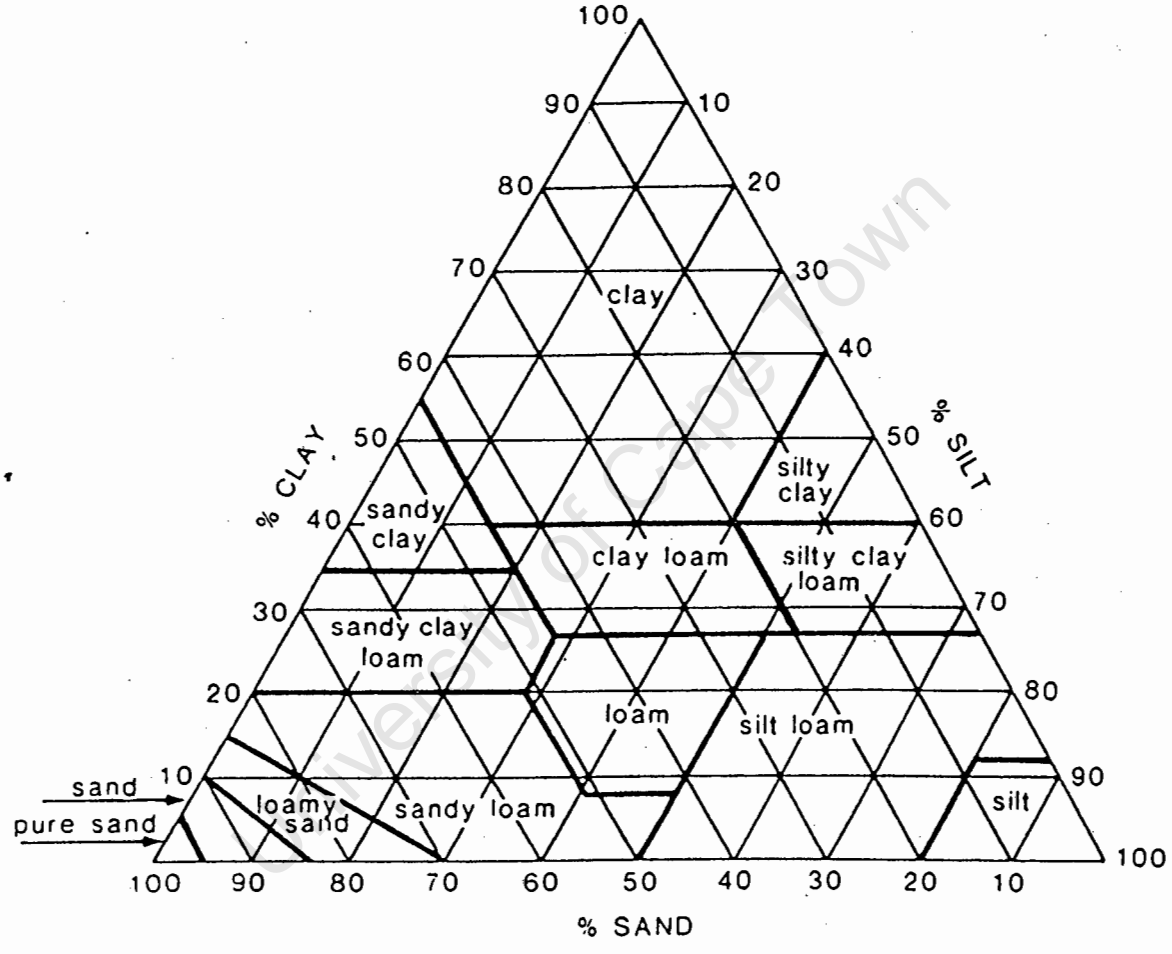

Figure II.1 Chart showing the percentages of clay (below 0.002 mm), silt (0.002 - 0.05 mm) and sand (0.05 - 2 mm) in the basic soil textural classes.

Table II.1 Landfill leachate analyses conducted by CSIR for Waste-tech on the 7 April 1995. The analyses indicate *inter alia* that calcium concentrations can reach values as high as 3197 $mg.l^{-1}$, almost certainly high enough to promote calcite precipitation at pH 7.7 and log P_{CO2} of -4.5.

LAB No.	4722	4723	4724	4725
Sample No.	1	2	3	4
Sample ID	Leachate 1	North sump	South sump	East sump
Sample date	950405	950405	950405	950405
METAL ($mg.l^{-1}$)				
K	2108	3236	3980	10900
Na	14163	9052	13152	5568
Ca	3197	45	2939	85
Mg	664	366	646	97
Cd	<0.1	<0.1	<0.1	<0.1
Cu	0.3	0.1	0.3	0.2
Cr	0.7	0.6	0.7	0.5
Mn	31	3.6	39	0.6
Ni	4.3	0.9	4.6	0.4
Pb	0.3	<0.1	0.3	<0.1
Zn	20	2.4	29	1.6
N	2140	287	2210	2080
Cl	19489	12035	18272	8615
Alkalinity (as $CaCO_3$)	12000	8350	13650	21650
COD	64000	5910	62000	7730
Dissolved organic C	20600	1830	19800	2480
EC ($mS.m^{-1}$) (Lab)	5800	3900	5800	4400
pH (Lab)	6.2	9.2	8.4	10.2
Total hardness as $CaCO_3$	10720	1621	9998	611
Charge Balance				
% difference	21.02	0.98	21.44	2.22
CATIONS $meq.l^{-1}$	1036.9	529.4	1031.4	681.7
ANIONS $meq.l^{-1}$	856.8	524.2	849.4	696.8

Recorded measurements and calculated hydraulic conductivity values made during leaching column work.
The positions where sodium carbonate and landfill leachate were introduced into the leaching columns are indica

Column A - 8% Kaolinite amended soil.

Replicate	Hours	Minutes	Seconds	Time (min)	Cum. Vol. (ml)	K (cm/s)	log K	Pore volum
sodium carbonate								
0		0	0	0	0			
1		1	59	1.98	11	2.30E-04	-3.64	0.22
2		2	49	2.82	15	1.99E-04	-3.70	0.30
3		3	54	3.90	20	1.92E-04	-3.72	0.40
4		5	5	5.08	25	1.75E-04	-3.76	0.50
5		6	15	6.25	30	1.78E-04	-3.75	0.60
6		7	35	7.58	35	1.56E-04	-3.81	0.70
7		8	49	8.82	40	1.68E-04	-3.77	0.80
8		10	10	10.17	45	1.54E-04	-3.81	0.90
9		11	27	11.45	50	1.62E-04	-3.79	1.01
10		14	14	14.23	60	1.49E-04	-3.83	1.21
11		15	38	15.63	65	1.48E-04	-3.83	1.31
12		16	29	16.48	70	2.44E-04	-3.61	1.41
13		17	6	17.10	73	2.02E-04	-3.69	1.51
14		19	9	19.15	86	2.63E-04	-3.58	1.81
15		19	55	19.92	91	2.71E-04	-3.57	1.91
16		20	40	20.67	95	2.21E-04	-3.65	2.01
17		24	7	24.12	115	2.41E-04	-3.62	2.41
18		28	59	28.98	140	2.13E-04	-3.67	2.91
19		29	58	29.97	145	2.11E-04	-3.68	3.02
20		35	13	35.22	170	1.98E-04	-3.70	3.52
21		40	20	40.33	195	2.03E-04	-3.69	4.02
22		45	51	45.85	220	1.88E-04	-3.73	4.52

Replicate	Hours	Minutes	Seconds	Time (min)	Cum. Vol. (ml)	K (cm/s)	log K	Pore volum
23		51	39	51.65	245	1.79E-04	-3.75	5.03
24		57	40	57.67	270	1.72E-04	-3.76	5.53
25	1	3	57	63.95	295	1.65E-04	-3.78	6.03
26	1	17	20	77.33	345	1.55E-04	-3.81	7.04
27	1	31	25	91.42	395	1.47E-04	-3.83	8.04
28	4	5	10	245.17	895	1.35E-04	-3.87	18.09
29	4	21	30	261.50	995	2.54E-04	-3.60	20.10
30	4	35	50	275.83	1095	2.90E-04	-3.54	22.11
31	4	48	50	288.83	1195	3.19E-04	-3.50	24.12
32		13	10	302.00	1294	3.12E-04	-3.51	26.11
33		32	15	321.08	1466	3.74E-04	-3.43	29.57
34	1	17	18	366.13	1929	4.27E-04	-3.37	38.87
35	1	50	23	399.22	2350	5.28E-04	-3.28	47.34
36	2	11	52	420.70	2670	6.18E-04	-3.21	53.77
37	2	48	18	457.13	3257	6.69E-04	-3.17	65.57
38	3	12	43	481.55	3677	7.14E-04	-3.15	74.01
39	4	2	39	531.48	4700	8.50E-04	-3.07	94.57
leachate								
40		7	28	538.95	4820	6.67E-04	-3.18	96.58
41		12	41	544.17	4920	7.96E-04	-3.10	98.59
42		21	9	552.63	5050	6.37E-04	-3.20	100.60
43		26	56	558.42	5150	7.18E-04	-3.14	102.61
44		32	58	564.45	5250	6.88E-04	-3.16	104.62
45		40	3	571.53	5360	6.44E-04	-3.19	106.83
46		46	23	577.87	5453	6.09E-04	-3.22	108.70
47		52	33	584.03	5550	6.53E-04	-3.19	110.65
48		59	9	590.63	5650	6.29E-04	-3.20	112.66
49	1	24	29	615.97	5985	5.49E-04	-3.26	119.40
50	2	50	19	701.80	6985	4.83E-04	-3.32	127.44

Replicate	Hours	Minutes	Seconds	Time (min)	Cum. Vol. (ml)	K (cm/s)	log K	Pore volum
51	3	26	39	738.13	7485	5.71E-04	-3.24	137.49
52	4	5	7	776.60	8051	6.11E-04	-3.21	148.86

Column B - 8% Kaolinite-amended soil.

sodium carbonate								
1		51	15	51.25	100	8.10E-05	-4.09	2.01
2	2	6	35	126.58	270	9.37E-05	-4.03	5.43
3	3	5	16	185.27	467	1.39E-04	-3.86	9.39
4	5	54	46	354.77	615	3.62E-05	-4.44	12.36
5	7	25	0	445.00	681	3.04E-05	-4.52	13.69
6	8	10	10	490.17	712	2.85E-05	-4.55	14.31
7	18	0	20	1080.33	1472	5.34E-05	-4.27	29.59
8	19	21	38	1161.63	1591	6.07E-05	-4.22	33.29
9	20	18	47	1218.78	1681	6.54E-05	-4.18	35.66
10	25	11	22	1511.37	1986	4.33E-05	-4.36	38.73
11	26	23	52	1583.87	2262	1.58E-04	-3.80	44.28
12	27	53	52	1673.87	2692	1.98E-04	-3.70	52.92
leachate								
13	28	2	28	1682.47	2792	4.83E-04	-3.32	54.93
14	28	10	7	1690.12	2892	5.42E-04	-3.27	56.94
15	28	17	32	1697.53	2992	5.60E-04	-3.25	58.95
16	28	24	52	1704.87	3092	5.66E-04	-3.25	60.96
17	28	31	54	1711.90	3192	5.90E-04	-3.23	62.97
18	28	38	57	1718.95	3292	5.89E-04	-3.23	64.98
19	28	45	52	1725.87	3392	6.00E-04	-3.22	66.99
20	28	52	54	1732.90	3492	5.90E-04	-3.23	69.00

Replicate	Hours	Minutes	Seconds	Time (min)	Cum. Vol. (ml)	K (cm/s)	log K	Pore volum
21	28	59	58	1739.97	3600	6.34E-04	-3.20	71.18
22	29	20	0	1760.00	3800	4.14E-04	-3.38	75.20
23	29	41	9	1781.15	4100	5.89E-04	-3.23	81.23
24	30	33	9	1833.15	4788	5.49E-04	-3.26	95.05
25	31	29	9	1889.15	5486	5.17E-04	-3.29	109.08
26	31	59	49	1919.82	5912	5.76E-04	-3.24	117.65
27	32	21	42	1941.70	6211	5.67E-04	-3.25	123.66
28	32	33	10	1953.17	6383	6.22E-04	-3.21	127.11
29	33	18	13	1998.22	6987	5.56E-04	-3.25	139.25
30	33	51	19	2031.32	7495	6.37E-04	-3.20	149.47
31	34	17	19	2057.32	7871	6.00E-04	-3.22	157.02

Column A - 8% Kaolinite-amended soil with 4% gypsum-treated middle layer.

sodium carbonate

0	0	0	0.00	0			
1		30	0.50	10	8.30E-04	-3.08	0.20
2	1	3	1.05	20	7.55E-04	-3.12	0.40
3	1	37	1.62	30	7.32E-04	-3.14	0.60
4	2	15	2.25	40	6.55E-04	-3.18	0.80
5	2	58	2.97	50	5.79E-04	-3.24	1.01
6	3	39	3.65	60	6.07E-04	-3.22	1.21
7	4	26	4.43	70	5.30E-04	-3.28	1.41
8	5	12	5.20	80	5.41E-04	-3.27	1.61
9	6	2	6.03	90	4.98E-04	-3.30	1.81
10	7	18	7.30	105	4.91E-04	-3.31	2.11
11	11	26	11.43	150	4.52E-04	-3.35	3.02
12	17	0	17.00	205	4.10E-04	-3.39	4.12
13	22	0	22.00	250	3.74E-04	-3.43	5.03

Replicate	Hours	Minutes	Seconds	Time (min)	Cum. Vol. (ml)	K (cm/s)	log K	Pore volum
14		27	57	27.95	300	3.49E-04	-3.46	6.03
15		35	51	35.85	362	3.26E-04	-3.49	7.28
16		42	57	42.95	412	2.92E-04	-3.53	8.28
17		50	35	50.58	462	2.72E-04	-3.57	9.29
18	1	52	27	112.45	782	2.15E-04	-3.67	15.72
19	2	36	12	156.20	935	4.91E-05	-4.31	18.79
20	5	5	48	305.80	1322	1.07E-04	-3.97	26.57
21	6	38	24	398.40	1496	7.80E-05	-4.11	30.07
22	7	22	28	442.47	1570	6.97E-05	-4.16	31.56
23	20	42	37	1242.62	2390	4.25E-05	-4.37	48.04
24	22	5	31	1325.52	2521	6.56E-05	-4.18	50.67
25	22	55	10	1375.17	2598	6.44E-05	-4.19	52.22
26	27	49	18	1669.30	3008	5.78E-05	-4.24	54.61
27	28	59	7	1739.12	3154	8.68E-05	-4.06	57.55
28	30	28	26	1828.43	3329	8.13E-05	-4.09	61.06
29	30	37	14	1837.23	3356	1.27E-04	-3.90	62.57
30	30	43	42	1843.70	3378	1.41E-04	-3.85	63.82
31	30	56	58	1856.97	3419	1.28E-04	-3.89	66.11
leachate								
32		11	5	1868.05	3452	1.24E-04	-3.91	67.07
33		41	23	1898.35	3552	1.37E-04	-3.86	69.08
34	1	3	30	1920.47	3616	1.20E-04	-3.92	70.37
35	1	48	0	1964.97	3722	9.89E-05	-4.01	72.50
36	3	7	40	2044.63	3919	1.03E-04	-3.99	76.46
37	5	47	20	2204.30	4238	8.29E-05	-4.08	82.87
38	8	57	40	2394.63	4553	6.87E-05	-4.16	89.21
39	11	40	14	2557.20	4863	7.91E-05	-4.10	95.44
40	14	13	29	2710.45	5164	8.15E-05	-4.09	101.49
41	18	14	14	2951.20	5670	8.72E-05	-4.06	112.06

Replicate	Hours	Minutes	Seconds	Time (min)	Cum. Vol. (ml)	K (cm/s)	log K	Pore volum
42	21	32	14	3149.20	5935	5.55E-05	-4.26	114.39
43	26	44	24	3461.37	6392	6.08E-05	-4.22	123.58
44	41	21	13	4338.18	7430	4.91E-05	-4.31	144.44
45	42	47	19	4424.28	7547	5.64E-05	-4.25	146.79

Column B - 8% Kaolinite-amended soil with 4% gypsum-treated middle layer.

sodium carbonate								
1		5	30	5.50	17	1.28E-04	-3.89	0.34
2		6	35	6.58	20	1.15E-04	-3.94	0.40
3		10	0	10.00	30	1.21E-04	-3.92	0.60
4		13	32	13.53	40	1.17E-04	-3.93	0.80
5		17	13	17.22	50	1.13E-04	-3.95	1.01
6		21	10	21.17	60	1.05E-04	-3.98	1.21
7		25	17	25.28	70	1.01E-04	-4.00	1.41
8		29	22	29.37	80	1.02E-04	-3.99	1.61
9		33	37	33.62	90	9.76E-05	-4.01	1.81
10		37	58	37.97	100	9.54E-05	-4.02	2.01
11	1	1	39	61.65	150	8.76E-05	-4.06	3.02
12	1	7	32	67.53	162	8.46E-05	-4.07	3.26
13	2	35	55	155.92	341	8.40E-05	-4.08	5.95
14	15	35	57	935.95	1291	5.05E-05	-4.30	13.57
15	15	45	18	945.30	1302	4.88E-05	-4.31	13.65
16	18	34	34	1114.57	1441	3.41E-05	-4.47	14.93
17	23	14	16	1394.27	1642	2.98E-05	-4.53	16.84
leachate								
18	23	29	56	1409.93	1656	3.71E-05	-4.43	17.13

Replicate	Hours	Minutes	Seconds	Time (min)	Cum. Vol. (ml)	K (cm/s)	log K	Pore volum
19	24	28	33	1468.55	1706	3.54E-05	-4.45	18.13
20	24	52	31	1492.52	1727	3.64E-05	-4.44	18.61
21	25	55	50	1555.83	1784	3.74E-05	-4.43	20.62
22	26	41	27	1601.45	1841	5.19E-05	-4.29	21.99
23	27	41	31	1661.52	1931	6.22E-05	-4.21	24.00
24	28	39	41	1719.68	2000	4.92E-05	-4.31	26.01
25	29	25	24	1765.40	2072	6.54E-05	-4.18	27.72
26	30	0	33	1800.55	2111	4.60E-05	-4.34	29.07
27	30	20	53	1820.88	2137	5.31E-05	-4.28	29.91
28	31	0	30	1860.50	2187	5.24E-05	-4.28	43.96
29	31	36	20	1896.33	2237	5.79E-05	-4.24	44.96
30	32	50	10	1970.17	2339	5.73E-05	-4.24	47.01
31	37	30	0	2250.00	2561	3.29E-05	-4.48	51.48
32	38	5	5	2285.08	2600	4.61E-05	-4.34	52.26
33	41	53	27	2513.45	2925	5.91E-05	-4.23	58.79
34	49	57	52	2997.87	3381	3.91E-05	-4.41	67.96
35	56	5	21	3365.35	3678	3.35E-05	-4.47	73.93
36	58	13	12	3493.20	3825	4.77E-05	-4.32	76.88
37	61	49	39	3709.65	3975	2.88E-05	-4.54	79.90
38	65	2	18	3902.30	4125	3.23E-05	-4.49	82.91
39		22	0	3924.30	4145	3.77E-05	-4.42	83.32
40	2	64	0	4086.30	4242	2.48E-05	-4.60	85.27
41	6	59	30	4321.80	4373	2.31E-05	-4.64	87.90
42	14	40	10	4782.47	4611	2.14E-05	-4.67	92.68
43	18	29	51	5012.15	4742	2.37E-05	-4.63	95.32
44	23	47	18	5329.60	4861	1.56E-05	-4.81	97.71
45	31	39	26	5801.73	5108	2.17E-05	-4.66	102.67
46	37	57	9	6179.45	5299	2.10E-05	-4.68	106.51
47	43	54	41	6536.98	5431	1.53E-05	-4.81	109.16
48	51	37	39	6999.95	5683	2.26E-05	-4.65	114.23

Replicate	Hours	Minutes	Seconds (min)	Time (ml)	Cum. Vol. (cm/s)	K	log K	ore volum
49	53	45	51	7128.15	5753	2.27E-05	-4.64	115.64
50	59	30	7	7472.42	5932	2.16E-05	-4.67	119.23
51	67	41	15	7963.55	6174	2.04E-05	-4.69	124.10
52	69	20	0	8062.30	6233	2.48E-05	-4.61	125.29
53	71	30		8192.30	6296	2.01E-05	-4.70	126.55
54	79	40		8682.30	6542	2.08E-05	-4.68	131.50
55	87	23		9145.30	6781	2.14E-05	-4.67	136.30
56	93	36		9518.30	6964	2.04E-05	-4.69	139.98
57	97	46		9768.30	7105	2.34E-05	-4.63	142.81
58	105	19		10221.30	7323	2.00E-05	-4.70	147.19
59	115	53		10855.30	7799	3.12E-05	-4.51	156.76
60	123	21		11303.30	8067	2.48E-05	-4.61	162.15

Column A - 8% Bentonite-amended soil.
Saturated with sodium carbonate.

leachate

1	47	30	0	2850.00	1.03	1.50E-08	-7.82	0.02
2	94	36	0	5676.00	2.3	1.87E-08	-7.73	0.05
3	240	26	0	14426.00	12.06	4.63E-08	-7.33	0.24
4	256	48	0	15408.00	222.06	8.87E-06	-5.05	4.46
5	260	15	0	15615.00	323.06	2.02E-05	-4.69	6.49
6	264	39	0	15879.00	445.06	1.92E-05	-4.72	8.95
7	269	56	0	16196.00	571.06	1.65E-05	-4.78	11.48
8	280	41	0	16841.00	781.06	1.35E-05	-4.87	15.70
9	289	30	0	17370.00	981.06	1.57E-05	-4.80	19.72

Column B - 8% Bentonite-amended soil
Saturated with sodium carbonate.

Replicate leachate	Hours	Minutes	Seconds (min)	Time (ml)	Cum. Vol. (cm/s)	K	log K	ore volum
1	47	30	0	2850.00	2.28	3.32E-08	-7.48	0.05
2	69	50	0	4190.00	16.28	1.61E-07	-6.79	0.33
3	82	53	0	4973.00	450.28	3.76E-06	-5.43	9.05
4	85	58	0	5158.00	518.28	4.17E-06	-5.38	10.42
5	87	48	0	5268.00	565.28	4.45E-06	-5.35	11.36
6	88	28	0	5308.00	623.28	4.87E-06	-5.31	12.53
7	89	53	0	5393.00	732.28	5.64E-06	-5.25	14.72
8	91	24	0	5484.00	849.28	6.43E-06	-5.19	17.07

Column A - Reproducibility test, unamended soil, leaching with distilled water.

Replicate	Hours	Minutes	Seconds	Time (min)	Cum. Vol. (ml)	K (cm/s)	log K	Pore volum
1			29	0.48	200	3.12E-02	-1.51	3.68
2			56	0.93	400	3.24E-02	-1.49	7.36
3		1	23	1.38	600	3.27E-02	-1.48	11.04
4		1	51	1.85	800	3.26E-02	-1.49	14.73
5		2	18	2.30	1000	3.28E-02	-1.48	18.41
6		2	47	2.78	1200	3.26E-02	-1.49	22.09
7		3	15	3.25	1400	3.25E-02	-1.49	25.77
8		3	43	3.72	1600	3.25E-02	-1.49	29.45
9		4	12	4.20	1800	3.24E-02	-1.49	33.13
10		4	41	4.68	2000	3.22E-02	-1.49	36.81
11		5	38	5.63	2400	3.22E-02	-1.49	44.18
12		6	36	6.60	2800	3.20E-02	-1.49	51.54
13		7	34	7.57	3200	3.19E-02	-1.50	58.90
14		8	32	8.53	3600	3.19E-02	-1.50	66.26
15		9	30	9.50	4000	3.18E-02	-1.50	73.63

Replicate	Hours	Minutes	Seconds	Time (min)	Cum. Vol. (ml)	K (cm/s)	log K	Pore volum
16		10	29	10.48	4400	3.17E-02	-1.50	80.99
17		11	27	11.45	4800	3.17E-02	-1.50	88.35
18		12	28	12.47	5200	3.15E-02	-1.50	95.71
19		13	28	13.47	5600	3.14E-02	-1.50	103.08
20		14	28	14.47	6000	3.13E-02	-1.50	110.44
21		15	28	15.47	6400	3.12E-02	-1.51	117.80
22		16	28	16.47	6800	3.12E-02	-1.51	125.17
23		17	29	17.48	7200	3.11E-02	-1.51	132.53
24		18	31	18.52	7600	3.10E-02	-1.51	139.89
25		19	32	19.53	8000	3.09E-02	-1.51	147.25

Column B - Reproducibility test, unamended soil, leaching with distilled water.

1			25	0.42	200	3.62E-02	-1.44	3.68
2			52	0.87	400	3.48E-02	-1.46	7.36
3		1	19	1.32	600	3.44E-02	-1.46	11.04
4		1	47	1.78	800	3.39E-02	-1.47	14.73
5		2	14	2.23	1000	3.38E-02	-1.47	18.41
6		2	43	2.72	1200	3.33E-02	-1.48	22.09
7		3	10	3.17	1400	3.34E-02	-1.48	25.77
8		3	37	3.62	1600	3.34E-02	-1.48	29.45
9		4	4	4.07	1800	3.34E-02	-1.48	33.13
10		4	34	4.57	2000	3.31E-02	-1.48	36.81
11		5	32	5.53	2400	3.27E-02	-1.48	44.18
12		6	29	6.48	2800	3.26E-02	-1.49	51.54
13		7	29	7.48	3200	3.23E-02	-1.49	58.90
14		8	28	8.47	3600	3.21E-02	-1.49	66.26
15		9	28	9.47	4000	3.19E-02	-1.50	73.63
16		10	30	10.50	4400	3.16E-02	-1.50	80.99
17		11	31	11.52	4800	3.15E-02	-1.50	88.35

Replicate	Hours	Minutes	Seconds	Time (min)	Cum. Vol. (ml)	K (cm/s)	log K	Pore volum
18		12	36	12.60	5200	3.12E-02	-1.51	95.71
19		13	41	13.68	5600	3.09E-02	-1.51	103.08
20		14	47	14.78	6000	3.06E-02	-1.51	110.44
21		15	55	15.92	6400	3.04E-02	-1.52	117.80
22		17	4	17.07	6800	3.01E-02	-1.52	125.17
23		18	13	18.22	7200	2.98E-02	-1.53	132.53
24		19	23	19.38	7600	2.96E-02	-1.53	139.89
25		20	35	20.58	8000	2.93E-02	-1.53	147.25

Appendix 4

LEACHW sample input file for a 1m soil cover depth and 90% vegetation cover used for 1970.
The file contains all rainfall, vegetation, evaporation and soil input data required for the simulation.

```
A10veg9  < DOS Filename, 8 characters with no extension. Used in batch runs (started as LEACHW<filename).
-------------------------------------------------------------------------
□□     LEACHW WATER FLOW DATA FILE.
All numeric data are in positions 1 to 78, comments may extend to position 120.
Unless defined as 'not read' a value must be present for each item, although it may not be used.
Free format with blank delimiters. Preserve division and heading records.  No. of depth segments may be changed.
*************************************************************************************************************
 1     <Date format (1: month/day,year;  2: day/month/year).  Dates must be 6 digits, 2 each for day, mo, yr.
 052770 <Starting date.  No date in the input data should precede this date.
 092570 <Ending date or day number.  The starting date is day 1. (A value <010101 is treated as a day number).
 0.05   <Largest time interval within a day (0.1 day or less).
 0.010  <Maximum water flux per time step. (Dimensionless: flux (mm)/segment thickness (mm).
 1      <Number of repetitions of rainfall, crop and chemical application data.
 1000   <Profile depth (mm), preferably a multiple of the segment thickness.
 100    <Segment thickness (mm).  (The number of segments should be between about 8 and 30.
 2      <Lower boundary condition: 1:fixed depth water table; 2:free drainage, 3:zero flux 4:lysimeter.
 1300   <If the lower boundary is 1 or 5: initial water table depth (mm).
*************************************************************************************************************
*************************************************************************************************************
 2      <Number of output files: 1: OUT only; 2: OUT + SUM; 3: OUT + SUM + BTC
---------------------------------------
--- For the *.OUT file :
 1      <Units for depth data: 1: ug/kg, 2: mg/m2 per segment.  (Not used in LEACHW)
 1      <Node print frequency (print data for every node (1), alternate nodes (2).
 2      <Print options: 1, 2 or 3.  To select one of the following 3 options.
 1      <Option 1: Time steps/print (these are not of equal length)
46.00   <Option 2: Print at fixed time intervals (days between prints)
 5      <Option 3: No. of prints (the times for which are specified below)
 2      <Tables printed: 1: mass balance; 2: + depth data; 3: + crop data
---------------------------------------
--- For the * .SUM file :
 1.00   <Summary print interval (d)
 000    <Surface to [depth 1?] mm   ( Three depth segments for the
 000    <Depth 1 to [depth 2?] mm     summary file. Zero defaults to nodes
 000    <Depth 2 to [depth 3?] mm     closest to thirds of the profile)
---------------------------------------
--- For the *.BTC (breakthrough) file :
 1.0    <Incremental depth of drainage water per output (mm)
-------------------------------------------------------------------------
-- List here the times at which the *.OUT file is desired for print option 3.
-- The number of records must match the 'No. of prints' under option 3 above.
□Date or   Time of day        (At least one must be specified
□Day no.  (to nearest tenth)     even if print option is not 3)
□--------  ----------------
□000001        .2       (These dates can be past the last day)
□000046        .2
□000092        .2
```

□000138 .2
□000184 .2
****
****
□□ SOIL PHYSICAL PROPERTIES

-- Retentivity model 0 uses listed Campbell's retention parameters, otherwise
-- the desired particle size-based regression model (Table 2.1 in manual) is used.

Soil layer no.	Clay %	Silt %	Organic carbon %	Retention model	Starting theta or pot'l (one is used) kPa	Roots (for no growth) (relative)	Starting temp (C) (not read in LEACHW,C)
1	0.05	0.05	.006	1	.000	-1500.	.10	20.
2	0.05	0.05	.006	1	.000	-1500.	.20	20.
3	0.05	0.05	.006	1	.000	-1500.	.20	20.
4	0.05	0.05	.006	1	.000	-1500.	.15	20.
5	0.05	0.05	.006	1	.000	-1500.	.10	20.
6	0.05	0.05	.006	1	.000	-1500.	.05	20.
7	0.05	0.05	.006	1	.000	-1500.	.05	20.
8	0.05	0.05	.006	1	.000	-1500.	.05	20.
9	0.05	0.05	.006	1	.000	-1500	.05	20.
10	0.05	0.05	.006	1	.000	-1500	.05	20.

---
2 < Use listed water contents (1) or potentials (2) as starting values.
Particle density: Clay Silt and sand Organic matter (kg/dm3) (to calculate porosity)
□□ 2.65 2.67 1.10
****
For a uniform profile: Any non-zero value here will override those in
the table below (only if retentivity model is 0).

0.0 <Soil bulk density (kg/dm3)
-0.00 <'Air-entry value' (AEV) (kPa) (a in eq 2.1 to 2.4).
0.00 <Exponent (BCAM) in Campbell's water retention equation (b in eq. 2.1 to 2.4).
0000 -0.0 <Conductivity (mm/day) and corresponding matric potential (kPa) (for potential-based version of eq. 2.5)
0.0 <Pore interaction parameter (P) in Campbell's conductivity equation (eq.2.5 in manual).
0 <Dispersivity (mm) (eq. 3.12). (Read, but not used in LEACHW)
****

Soil segment no.	Soil retentivity parameters AEV kPa	BCAM	Bulk density kg/dm3	Match K(h) curve at: K mm/d	Matric pot'l kPa	using P	Dispersivity (not read in LEACHW) mm
1	-5.00	3.00	1.696	2590	-00.	1.0	100.
2	-5.00	3.00	1.696	2590	-00.	1.0	100.
3	-5.00	3.00	1.696	2590	-00.	1.0	100.
4	-5.00	3.00	1.696	2590	-00.	1.0	100.
5	-5.00	3.00	1.696	2590	-00.	1.0	100.
6	-5.00	3.00	1.696	2590	-00.	1.0	100.
7	-5.00	3.00	1.696	2590	-00.	1.0	100.
8	-5.00	3.00	1.696	2590	-00.	1.0	100.
9	-5.00	3.00	1.696	2590	-00.	1.0	100.
10	-5.00	3.00	1.696	2590	-00.	1.0	100.

****
****

□□□ CROP DATA

Data for at least one crop must be specified, even if no crop desired.
For fallow soil, set flag below to 0, or germination past the simulation end date.

---

1 <Plants present: 1 yes, 0 no. This flag overrides all other crop data.
1 <No. of crops (>0), even if bypassed. Dates can be past last day of simulation.
1 <Growth: 1:No(use root data specified above, crop cover below); 2:Yes.
-1500 <Wilting point (soil) kPa.
-3000 <Min.root water pot'l(kpa).
1.1 <Maximum ratio of actual to potential transpiration (dry surface).
1.05 <Root resistance (weights water uptake by depth). (>1, No weighting: 1.0). See Eq. 2-16.

Crop no	Germination	Emergence Root	Maturity Plant	Harvest	Rel. root depth	Crop cover	Pan factor	Annual N uptake kg/ha	(read in LEACHN only)
	..... Date or Day no								
1	052770	052870	052870	052870	121270	1.00	0.9	1.00	102

****

****

□□ RAIN/IRRIGATION AND WATER COMPOSITION

□□ ---

Choosing the steady-state flow option will prevent calls to the water flow subroutine, fix fluxes and concentrations at those specified below and maintain theta constant with time. Interrupted steady-state can be specified by appropriate times and amounts. For the steady-state flow option, use a uniform soil column.

---

1 < Water flow: Richards (1), modified Addiscott (2), steady-state (3).
-5 < For Addiscott : matric potential at field capacity (kPa).
-200 < : division between mobile and immobile water (kPa).
0.4 < For steady-state: Water content in uniform column (theta)
60 < Number of water applications. Some or all can be past last day.

Start time Date or Day no.	Time of day	Amount mm	Surface flux density mm/d	Dissolved in water (can be 0) Chem1 (not read or used in LEACHW)	Chem2	Chem3	Chem4.....
000001	.2	16.00	16.00	00	00	00	00
000002	.2	6.10	6.10	00	00	00	00
000003	.2	1.10	1.10	00	00	00	00
000007	.2	.90	.90	00	00	00	00
000008	.2	0.30	0.30	00	00	00	00
000009	.2	2.20	2.20	00	00	00	00
000010	.2	18.20	18.20	00	00	00	00
000011	.2	11.1	11.10	00	00	00	00
000014	.2	37.5	37.50	00	00	00	00
000015	.2	3.0	3.00	00	00	00	00
000016	.2	5.5	5.50	00	00	00	00
000020	.2	1.0	1.00	00	00	00	00
000021	.2	0.2	0.20	00	00	00	00
000022	.2	3.4	3.40	00	00	00	00
000023	.2	3.5	3.50	00	00	00	00
000027	.2	26.5	26.50	00	00	00	00

000028	.2	7.0	7.00	00	00	00	00
000029	.2	1.4	1.40	00	00	00	00
000033	.2	17.5	17.50	00	00	00	00
000034	.2	3.9	3.90	00	00	00	00
000037	.2	0.3	0.30	00	00	00	00
000038	.2	1.20	1.20	00	00	00	00
000039	.2	5.00	5.00	00	00	00	00
000043	.2	3.80	3.80	00	00	00	00
000044	.2	0.40	0.40	00	00	00	00
000045	.2	0.30	0.30	00	00	00	00
000046	.2	6.50	6.50	00	00	00	00
000047	.2	0.20	0.20	00	00	00	00
000051	.2	23.60	23.60	00	00	00	00
000052	.2	1.00	1.00	00	00	00	00
000054	.2	39.50	39.50	00	00	00	00
000055	.2	9.50	9.50	00	00	00	00
000060	.2	11.90	11.90	00	00	00	00
000061	.2	17.00	17.00	00	00	00	00
000062	.2	21.00	21.00	00	00	00	00
000063	.2	3.00	3.00	00	00	00	00
000067	.2	4.50	4.50	00	00	00	00
000070	.2	6.00	6.00	00	00	00	00
000072	.2	3.70	3.70	00	00	00	00
000073	.2	1.10	1.10	00	00	00	00
000079	.2	4.10	4.10	00	00	00	00
000085	.2	10.90	10.90	00	00	00	00
000087	.2	31.00	31.00	00	00	00	00
000088	.2	1.50	1.50	00	00	00	00
000089	.2	4.50	4.50	00	00	00	00
000090	.2	8.50	8.50	00	00	00	00
000091	.2	2.20	2.20	00	00	00	00
000096	.2	4.50	4.5	00	00	00	00
000097	.2	4.2	4.2	00	0	0	0
000098	.2	3.4	3.4	0	0	0	0
000099	.2	5.5	5.5	0	0	0	0
000101	.2	2.2	2.2	0	0	0	0
000102	.2	2.0	2.0	00	00	00	00
000105	.2	19.2	19.2	0	0	0	0
000106	.2	8.4	8.4	0	0	0	0
000107	.2	0.5	0.5	0	0	0	0
000110	.2	0.5	0.5	0	0	0	0
000115	.2	1.5	1.5	0	0	0	0
000121	.2	14.0	14.0	0	0	0	0
000122	.2	5.0	5.0	0	0	0	0

****

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□ POTENTIAL ET (WEEKLY TOTALS, mm), DEPTH TO WATER TABLE (mm)
□ MEAN WEEKLY TEMPERATURES AND MEAN WEEKLY AMPLITUDE (degrees C)

	Week no.	ET	Water table	Mean temp	Amplitude
□1	48.5	0.	19.5	6.9	
□2	41.2	0.	16.2	4.4	
□3	47.6	0.	15.4	5.9	

□4	40.7	0.	12.8	5.8
□5	44.4	0.	13.1	5.8
□6	38.4	0.	14.8	5.6
□7	40.6	0.	20.1	3.9
□8	35.0	0.	17.4	5.2
□9	39.2	0.	16.7	5.6
10	31.8	0.	17.6	6.3
11	39.3	0.	18.5	7.7
12	41.2	0.	21.7	6.1
13	22.6	0.	23.8	3.9
14	24.0	0.	26.2	5.3
15	17.5	0.	22.4	6.6
16	13.7	0.	23.9	4.6
17	12.0	0.	26.0	5.1
18	6.7	0.	20.7	6.2
19	7.8	0.	26.1	3.4
20	5.9	0.	25.6	5.8
21	7.1	0.	28.1	6.1
22	6.7	0.	27.2	4.2
23	6.2	0.	27.8	5.6
24	8.4	0.	28.3	5.6
25	13.0	0.	28.6	5.7
26	7.9	0.	26.7	5.6
27	2.7	0.	27.7	5.5
28	7.1	0.	28.0	5.6
29	4.0	0.	28.8	5.7
30	6.6	0.	28.7	5.6
31	7.6	0.	27.3	5.0
32	10.5	0.	26.2	5.8
33	10.0	0.	26.4	6.5
34	10.1	0.	26.6	4.2
35	10.9	0.	19.4	5.6
36	11.7	0.	24.5	4.7
37	12.9	0.	24.8	3.9
38	10.2	0.	23.9	3.7
39	21.0	0.	23.1	3.2
40	20.1	0.	0	0
41	21.5	0.	0	0
42	27.9	0.	0	0
43	25.6	0.	0	0
44	31.3	0.	0	0
45	30.9	0.	0	0
46	38.9	0.	0	0
47	30.3	0.	0	0
48	42.5	0.	0	0
49	41.8	0.	0	0
50	37.8	0.	0	0
51	31.2	0.	0	0
52	45.9	0.	0	0

LEACHW sample input file for a 1m soil cover depth and 90% vegetation cover used for 1954.
The file contains all rainfall, vegetation, evaporation and soil input data required for the simulation.

```
w10veg9  < DOS Filename. 8 characters with no extension. Used in batch runs (started as LEACHW<filename).
-------------------------------------------------------------------------
□□     LEACHW WATER FLOW DATA FILE.
All numeric data are in positions 1 to 78. comments may extend to position 120.
Unless defined as 'not read' a value must be present for each item, although it may not be used.
Free format with blank delimiters. Preserve division and heading records.  No. of depth segments may be changed.
*************************************************************************************************************
 1     <Date format (1: month/day,year;  2: day/month/year).  Dates must be 6 digits, 2 each for day, mo, yr.
 051554 <Starting date.  No date in the input data should precede this date.
 091954 <Ending date or day number.  The starting date is day 1. (A value <010101 is treated as a day number).
 0.05   <Largest time interval within a day (0.1 day or less).
 0.010  <Maximum water flux per time step. (Dimensionless: flux (mm)/segment thickness (mm).
 1      <Number of repetitions of rainfall, crop and chemical application data.
 1000   <Profile depth (mm), preferably a multiple of the segment thickness.
 100    <Segment thickness (mm).  (The number of segments should be between about 8 and 30.
 2      <Lower boundary condition: 1:fixed depth water table; 2:free drainage, 3:zero flux 4:lysimeter.
 1300   <If the lower boundary is 1 or 5: initial water table depth (mm).
*************************************************************************************************************
*************************************************************************************************************
 2      <Number of output files: 1: OUT only; 2: OUT + SUM; 3: OUT + SUM + BTC
---------------------------------------
--- For the *.OUT file :
 1      <Units for depth data: 1: ug/kg, 2: mg/m2 per segment.  (Not used in LEACHW)
 1      <Node print frequency (print data for every node (1), alternate nodes (2).
 2      <Print options: 1, 2 or 3.  To select one of the following 3 options.
 1      <Option 1: Time steps/print (these are not of equal length)
20.00   <Option 2: Print at fixed time intervals (days between prints)
 7      <Option 3: No. of prints (the times for which are specified below)
 2      <Tables printed: 1: mass balance; 2: + depth data; 3: + crop data
---------------------------------------
--- For the * .SUM file :
 1.00   <Summary print interval (d)
 300    <Surface to [depth 1?] mm     (Three depth segments for the
 600    <Depth 1 to [depth 2?] mm      summary file. Zero defaults to nodes
 1000   <Depth 2 to [depth 3?] mm      closest to thirds of the profile)
---------------------------------------
--- For the *.BTC (breakthrough) file :
 1.0    <Incremental depth of drainage water per output (mm)
-------------------------------------------------------------------------
-- List here the times at which the *.OUT file is desired for print option 3.
-- The number of records must match the 'No. of prints' under option 3 above.
□Date or    Time of day        (At least one must be specified
□Day no.  (to nearest tenth)     even if print option is not 3)
□--------  ---------------
□000001       .2       (These dates can be past the last day)
□000020       .2
□000040       .2
□000060       .2
□000080       .2
```

□000100 .2
□000120 .2
****
****
□□ SOIL PHYSICAL PROPERTIES

-- Retentivity model 0 uses listed Campbell's retention parameters, otherwise
-- the desired particle size-based regression model (Table 2.1 in manual) is used.

Soil layer no.	Clay %	Silt %	Organic carbon %	Retention model (one is used)	Starting theta or pot'l kPa	Roots (for no growth) (relative)	Starting temp (C) (not read in LEACHW,C)
1	0.05	0.05	.006	1	.000	-1500.	.10	20.
2	0.05	0.05	.006	1	.000	-1500.	.20	20.
3	0.05	0.05	.006	1	.000	-1500.	.20	20.
4	0.05	0.05	.006	1	.000	-1500.	.15	20.
5	0.05	0.05	.006	1	.000	-1500.	.10	20.
6	0.05	0.05	.006	1	.000	-1500.	.05	20.
7	0.05	0.05	.006	1	.000	-1500.	.05	20.
8	0.05	0.05	.006	1	.000	-1500.	.05	20.
9	0.05	0.05	.006	1	.000	-1500	.05	20.
10	0.05	0.05	.006	1	.000	-1500	.05	20.

---
2 < Use listed water contents (1) or potentials (2) as starting values.
Particle density: Clay Silt and sand Organic matter (kg/dm3) (to calculate porosity)
□□ 2.65 2.67 1.10
****
For a uniform profile: Any non-zero value here will override those in
the table below (only if retentivity model is 0).

0.0 <Soil bulk density (kg/dm3)
-0.00 <'Air-entry value' (AEV) (kPa) (a in eq 2.1 to 2.4).
0.00 <Exponent (BCAM) in Campbell's water retention equation (b in eq. 2.1 to 2.4).
0000 -0.0 <Conductivity (mm/day) and corresponding matric potential (kPa) (for potential-based version of eq. 2.5)
0.0 <Pore interaction parameter (P) in Campbell's conductivity equation (eq.2.5 in manual).
0 <Dispersivity (mm) (eq. 3.12). (Read, but not used in LEACHW)
****

Soil segment no.	Soil retentivity parameters AEV kPa	BCAM	Bulk density kg/dm3	Match K(h) curve at: K mm/d	Matric pot'l kPa	using P	Dispersivity (not read in LEACHW) mm
1	-5.00	3.00	1.696	2590	-00.	1.0	100.
2	-5.00	3.00	1.696	2590	-00.	1.0	100.
3	-5.00	3.00	1.696	2590	-00.	1.0	100.
4	-5.00	3.00	1.696	2590	-00.	1.0	100.
5	-5.00	3.00	1.696	2590	-00.	1.0	100.
6	-5.00	3.00	1.696	2590	-00.	1.0	100.
7	-5.00	3.00	1.696	2590	-00.	1.0	100.
8	-5.00	3.00	1.696	2590	-00.	1.0	100.
9	-5.00	3.00	1.696	2590	-00.	1.0	100.
10	-5.00	3.00	1.696	2590	-00.	1.0	100.

****
****

□□□ CROP DATA

Data for at least one crop must be specified, even if no crop desired.
For fallow soil, set flag below to 0, or germination past the simulation end date.

---

1 <Plants present: 1 yes, 0 no. This flag overrides all other crop data.
1 <No. of crops (>0), even if bypassed. Dates can be past last day of simulation.
1 <Growth: 1:No(use root data specified above, crop cover below); 2:Yes.
-1500 <Wilting point (soil) kPa.
-3000 <Min.root water pot'l(kpa).
1.1 <Maximum ratio of actual to potential transpiration (dry surface).
1.05 <Root resistance (weights water uptake by depth). (>1, No weighting: 1.0). See Eq. 2-16.

Crop no	Germination	Emergence Root	Maturity Plant	Harvest root depth	Rel. cover	Crop factor	Pan fraction	Annual N uptake kg/ha	(read in LEACHN only)
	..... Date or Day no								
1	051554	051654	051654	051654	121254	1.00	0.9	1.00	102

****

****

□□ RAIN/IRRIGATION AND WATER COMPOSITION

□□ ---

Choosing the steady-state flow option will prevent calls to the water flow subroutine, fix fluxes and concentrations at those specified below and maintain theta constant with time. Interrupted steady-state can be specified by appropriate times and amounts. For the steady-state flow option, use a uniform soil column.

---

1 < Water flow: Richards (1), modified Addiscott (2), steady-state (3).
-5 < For Addiscott : matric potential at field capacity (kPa).
-200 < : division between mobile and immobile water (kPa).
0.4 < For steady-state: Water content in uniform column (theta)
60 < Number of water applications. Some or all can be past last day.

Start time Date or Day no.	Time of day	Amount mm	Surface flux density mm/d	Dissolved in water (can be 0) Chem1 (not read or used in LEACHW)	Chem2	Chem3	Chem4.....
000001	.2	35.20	35.20	00	00	00	00
000002	.2	71.00	71.00	00	00	00	00
000003	.2	29.50	29.50	00	00	00	00
000004	.2	3.60	3.60	00	00	00	00
000006	.2	5.90	5.90	00	00	00	00
000007	.2	.20	.20	00	00	00	00
000009	.2	8.30	8.30	00	00	00	00
000010	.2	.20	.20	00	00	00	00
000012	.2	21.60	21.60	00	00	00	00
000013	.2	28.90	28.90	00	00	00	00
000014	.2	5.00	5.00	00	00	00	00
000015	.2	1.20	1.20	00	00	00	00
000016	.2	.40	.40	00	00	00	00
000021	.2	.40	.40	00	00	00	00
000030	.2	12.80	12.80	00	00	00	00
000031	.2	7.60	7.60	00	00	00	00

000037	.2	.30	.30	00	00	00	00
000038	.2	32.20	32.20	00	00	00	00
000039	.2	12.30	12.30	00	00	00	00
000040	.2	80.40	80.40	00	00	00	00
000041	.2	3.9	3.90	00	00	00	00
000051	.2	9.9	9.90	00	00	00	00
000052	.2	65.1	65.10	00	00	00	00
000053	.2	5.9	5.90	00	00	00	00
000054	.2	23 2	23.20	00	00	00	00
000055	.2	2.4	2.40	00	00	00	00
000056	.2	7.0	.70	00	00	00	00
000057	.2	0.4	.40	00	00	00	00
000059	.2	0.3	.30	00	00	00	00
000061	.2	31.5	31.50	00	00	00	00
000062	.2	3.0	3.00	00	00	00	00
000063	.2	40.6	40.60	00	00	00	00
000064	.2	5.8	5.80	00	00	00	00
000065	.2	30.3	30.30	00	00	00	00
000066	.2	12.80	12.80	00	00	00	00
000069	.2	10.90	10.90	00	00	00	00
000070	.2	.70	.70	00	00	00	00
000071	.2	4.40	4.40	00	00	00	00
000072	.2	2.30	2.30	00	00	00	00
000076	.2	26.00	26.00	00	00	00	00
000078	.2	6.50	6.50	00	00	00	00
000079	.2	27.30	27.30	00	00	00	00
000080	.2	.70	.70	00	00	00	00
000082	.2	1.10	1.10	00	00	00	00
000086	.2	10.70	10.70	00	00	00	00
000091	.2	.40	.40	00	00	00	00
000095	.2	13.80	13.80	00	00	00	00
000096	.2	6.30	6.30	00	00	00	00
000097	.2	32.80	32.80	00	00	00	00
000098	.2	8.10	8.10	00	00	00	00
000101	.2	.50	.50	00	00	00	00
000102	.2	3.00	3.00	00	00	00	00
000103	.2	.10	.10	00	00	00	00
000108	.2	18.20	18.20	00	00	00	00
000109	.2	14.30	14.30	00	00	00	00
000110	.2	1.20	1.20	00	00	00	00
000112	.2	3.90	3.90	00	00	00	00
000115	.2	14.70	14.70	00	00	00	00
000120	.2	3.30	3.30	00	00	00	00
000127	.2	.40	.40	00	00	00	00

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□ POTENTIAL ET (WEEKLY TOTALS, mm), DEPTH TO WATER TABLE (mm)
□ MEAN WEEKLY TEMPERATURES AND MEAN WEEKLY AMPLITUDE (degrees C)

Week no.	ET	Water table	Mean temp	Amplitude
□1	48.5	0.	19.5	6.9
□2	41.2	0.	16.2	4.4
□3	47.6	0.	15.4	5.9

□4	40.7	0.	12.8	5.8
□5	44.4	0.	13.1	5.8
□6	38.4	0.	14.8	5.6
□7	40.6	0.	20.1	3.9
□8	35.0	0.	17.4	5.2
□9	39.2	0.	16.7	5.6
10	31.8	0.	17.6	6.3
11	39.3	0.	18.5	7.7
12	41.2	0.	21.7	6.1
13	22.6	0.	23.8	3.9
14	24.0	0.	26.2	5.3
15	17.5	0.	22.4	6.6
16	13.7	0.	23.9	4.6
17	12.0	0.	26.0	5.1
18	6.7	0.	20.7	6.2
19	7.8	0.	26.1	3.4
20	5.9	0.	25.6	5.8
21	7.1	0.	28.1	6.1
22	6.7	0.	27.2	4.2
23	6.2	0.	27.8	5.6
24	8.4	0.	28.3	5.6
25	13.0	0.	28.6	5.7
26	7.9	0.	26.7	5.6
27	2.7	0.	27.7	5.5
28	7.1	0.	28.0	5.6
29	4.0	0.	28.8	5.7
30	6.6	0.	28.7	5.6
31	7.6	0.	27.3	5.0
32	10.5	0.	26.2	5.8
33	10.0	0.	26.4	6.5
34	10.1	0.	26.6	4.2
35	10.9	0.	19.4	5.6
36	11.7	0.	24.5	4.7
37	12.9	0.	24.8	3.9
38	10.2	0.	23.9	3.7
39	21.0	0.	23.1	3.2
40	20.1	0.	0	0
41	21.5	0.	0	0
42	27.9	0.	0	0
43	25.6	0.	0	0
44	31.3	0.	0	0
45	30.9	0.	0	0
46	38.9	0.	0	0
47	30.3	0.	0	0
48	42.5	0.	0	0
49	41.8	0.	0	0
50	37.8	0.	0	0
51	31.2	0.	0	0
52	45.9	0.	0	0

Appendix 5

LEACHW sample output file for 1m soil cover depth and 90% vegetation cover for 1970.

a10veg9 .OUT

SOIL HYDROLOGICAL CHARACTERISTICS

PREDICTED RETENTIVITY AND CONDUCTIVITY DATA

Depth (mm) | Water content, theta (Conductivity mm/day)

Satrn	-3 kPa	-10 kPa	-30 kPa	-100 kPa	-1500 kPa	a (kPa)	b	p
50.	.365	.233	.180	.143	.110	.062	-.371	4.683	1.00
	.259E+04	.104E+02	.433E+00	.238E-01	.990E-03	.776E-06			
150.	.365	.233	.180	.143	.110	.062	-.371	4.683	1.00
	.259E+04	.104E+02	.433E+00	.238E-01	.990E-03	.776E-06			
250.	.365	.233	.180	.143	.110	.062	-.371	4.683	1.00
	.259E+04	.104E+02	.433E+00	.238E-01	.990E-03	.776E-06			
350.	.365	.233	.180	.143	.110	.062	-.371	4.683	1.00
	.259E+04	.104E+02	.433E+00	.238E-01	.990E-03	.776E-06			
450.	.365	.233	.180	.143	.110	.062	-.371	4.683	1.00
	.259E+04	.104E+02	.433E+00	.238E-01	.990E-03	.776E-06			
550.	.365	.233	.180	.143	.110	.062	-.371	4.683	1.00
	.259E+04	.104E+02	.433E+00	.238E-01	.990E-03	.776E-06			
650.	.365	.233	.180	.143	.110	.062	-.371	4.683	1.00
	.259E+04	.104E+02	.433E+00	.238E-01	.990E-03	.776E-06			
750.	.365	.233	.180	.143	.110	.062	-.371	4.683	1.00
	.259E+04	.104E+02	.433E+00	.238E-01	.990E-03	.776E-06			
850.	.365	.233	.180	.143	.110	.062	-.371	4.683	1.00
	.259E+04	.104E+02	.433E+00	.238E-01	.990E-03	.776E-06			
950.	.365	.233	.180	.143	.110	.062	-.371	4.683	1.00
	.259E+04	.104E+02	.433E+00	.238E-01	.990E-03	.776E-06			

LEACHW used the Richards equation.

Day 1. Date 5/27/70

Day 46. Date 7/11/70

Time 0h 0 .0000000Day

Time24h 0 1.0000000Day

Cum. infiltration: .0	Cum. infiltration: 182.5
Could not infiltrate: .0	Could not infiltrate: .0
Cum. leachate : .0	Cum. leachate : .0
Cum. evaporation : .0	Cum. evaporation : 13.1
Ponded water : .0	Ponded water : 1.3
Cum. transpiration : .0	Cum. transpiration : 113.0
Prof. water change: .0	Prof. water change: 56.0
Profile water cont.: 61.9	Profile water cont.: 117.9
Crop cover fraction: .9000	Crop cover fraction: .9000
Mass balance error: .0	Mass balance error: .5

Depth mm	Theta	Potl. kPa	Flux mm	Root distrib	Depth mm	Theta	Potl. kPa	Flux mm	Root distrib
50	.062	-1500.0	.00	.100	50	.174	-12.0	169.44	.100
150	.062	-1500.0	.00	.200	150	.138	-35.3	127.09	.200
250	.062	-1500.0	.00	.200	250	.124	-58.3	80.13	.200
350	.062	-1500.0	.00	.150	350	.124	-58.0	50.11	.150
450	.062	-1500.0	.00	.100	450	.127	-52.1	31.37	.100
550	.062	-1500.0	.00	.050	550	.128	-50.1	19.53	.050
650	.062	-1500.0	.00	.050	650	.121	-64.2	11.89	.050
750	.062	-1500.0	.00	.050	750	.104	-129.8	5.78	.050
850	.062	-1500.0	.00	.050	850	.077	-542.4	1.53	.050
950	.062	-1500.0	.00	.050	950	.062	-1471.8	.03	.050
Drainage flux : .00					Drainage flux : .00				

(Water fluxes are cumulative since the previous printout and, except for the drainage flux, refer to the upper boundary of each depth segment.

Day 92.	Day 122.
Date 8/26/70	Date 9/25/70
Time24h 0 1.0000000Day	Time24h 0 1.0000000Day

Cum. infiltration: 388.5	Cum. infiltration: 458.4
Could not infiltrate: .0	Could not infiltrate: .0
Cum. leachate : 5.4	Cum. leachate : 24.7
Cum. evaporation : 27.2	Cum. evaporation : 30.9
Ponded water : .0	Ponded water : 1.0
Cum. transpiration : 240.2	Cum. transpiration : 273.8
Prof. water change: 114.8	Prof. water change: 128.1
Profile water cont.: 176.7	Profile water cont.: 190.0
Crop cover fraction: .9000	Crop cover fraction: .9000
Mass balance error: .9	Mass balance error: .9

Depth mm	Theta	Potl. kPa	Flux mm	Root distrib	Depth mm	Theta	Potl. kPa	Flux mm	Root distrib
50	.191	-7.8	191.86	.100	50	.216	-4.3	66.18	.100
150	.194	-7.1	167.41	.200	150	.212	-4.7	57.57	.200
250	.198	-6.4	130.63	.200	250	.205	-5.5	45.31	.200
350	.200	-6.1	101.53	.150	350	.193	-7.3	36.92	.150
450	.198	-6.4	78.82	.100	450	.181	-9.9	32.88	.100
550	.190	-7.8	60.09	.050	550	.176	-11.2	32.10	.050
650	.175	-11.5	47.65	.050	650	.177	-11.0	32.56	.050
750	.153	-21.9	36.90	.050	750	.179	-10.5	31.79	.050
850	.135	-38.7	27.25	.050	850	.180	-10.1	28.87	.050
950	.132	-43.0	16.81	.050	950	.181	-9.8	24.24	.050
Drainage flux :			5.36		Drainage flux :			19.34	

(Water fluxes are cumulative since the previous printout and, except for the drainage flux, refer to the upper boundary of each depth segment.

LEACHW sample output file for a 1m soil cover depth and 90% vegetation cover for 1954.

w10veg9 .OUT

SOIL HYDROLOGICAL CHARACTERISTICS

PREDICTED RETENTIVITY AND CONDUCTIVITY DATA

Depth (mm)	Water content, theta (Conductivity mm/day)								
	Satm	-3 kPa	-10 kPa	-30 kPa	-100 kPa	-1500 kPa	a (kPa)	b	p
50.	.365	.233	.180	.143	.110	.062	-.371	4.683	1.00
	.259E+04	.104E+02	.433E+00	.238E-01	.990E-03	.776E-06			
150.	.365	.233	.180	.143	.110	.062	-.371	4.683	1.00
	.259E+04	.104E+02	.433E+00	.238E-01	.990E-03	.776E-06			
250.	.365	.233	.180	.143	.110	.062	-.371	4.683	1.00
	.259E+04	.104E+02	.433E+00	.238E-01	.990E-03	.776E-06			
350.	.365	.233	.180	.143	.110	.062	-.371	4.683	1.00
	.259E+04	.104E+02	.433E+00	.238E-01	.990E-03	.776E-06			
450.	.365	.233	.180	.143	.110	.062	-.371	4.683	1.00
	.259E+04	.104E+02	.433E+00	.238E-01	.990E-03	.776E-06			
550.	.365	.233	.180	.143	.110	.062	-.371	4.683	1.00
	.259E+04	.104E+02	.433E+00	.238E-01	.990E-03	.776E-06			
650.	.365	.233	.180	.143	.110	.062	-.371	4.683	1.00
	.259E+04	.104E+02	.433E+00	.238E-01	.990E-03	.776E-06			
750.	.365	.233	.180	.143	.110	.062	-.371	4.683	1.00
	.259E+04	.104E+02	.433E+00	.238E-01	.990E-03	.776E-06			
850.	.365	.233	.180	.143	.110	.062	-.371	4.683	1.00
	.259E+04	.104E+02	.433E+00	.238E-01	.990E-03	.776E-06			
950.	.365	.233	.180	.143	.110	.062	-.371	4.683	1.00
	.259E+04	.104E+02	.433E+00	.238E-01	.990E-03	.776E-06			

LEACHW used the Richards equation.

Day 1.	Day 20.
Date 5/15/54	Date 6/ 3/54
Time 0h 0 .0000000Day	Time24h 0 1.0000000Day
Cum. infiltration: .0	Cum. infiltration: 211.0
Could not infiltrate: .0	Could not infiltrate: .0
Cum. leachate : .0	Cum. leachate : 37.8
Cum. evaporation : .0	Cum. evaporation : 6.0
Ponded water : .0	Ponded water : .0
Cum. transpiration : .0	Cum. transpiration : 53.9
Prof. water change: .0	Prof. water change: 112.5

Profile water cont.: 61.9	Profile water cont.: 174.4
Crop cover fraction: .9000	Crop cover fraction: .9000
Mass balance error: .0	Mass balance error: .9

Depth mm	Theta	Potl. kPa	Flux mm	Root distrib
50	.062	-1500.0	.00	.100
150	.062	-1500.0	.00	.200
250	.062	-1500.0	.00	.200
350	.062	-1500.0	.00	.150
450	.062	-1500.0	.00	.100
550	.062	-1500.0	.00	.050
650	.062	-1500.0	.00	.050
750	.062	-1500.0	.00	.050
850	.062	-1500.0	.00	.050
950	.062	-1500.0	.00	.050
Drainage flux :			.00	

Depth mm	Theta	Potl. kPa	Flux mm	Root distrib
50	.158	-18.6	205.02	.100
150	.163	-16.3	188.47	.200
250	.165	-15.1	164.55	.200
350	.169	-13.7	142.80	.150
450	.173	-12.2	124.44	.100
550	.178	-10.8	108.31	.050
650	.181	-9.8	94.00	.050
750	.184	-9.2	79.68	.050
850	.186	-8.6	65.44	.050
950	.188	-8.3	51.40	.050
Drainage flux :			37.77	

(Water fluxes are cumulative since the previous printout and, except for the drainage flux, refer to the upper boundary of each depth segment.

Day 40.	Day 60.
Date 6/23/54	Date 7/13/54
Time24h 0 1.0000000Day	Time24h 0 1.0000000Day
Cum. infiltration: 340.9	Cum. infiltration: 447.9
Could not infiltrate: .0	Could not infiltrate: 27.0
Cum. leachate : 39.6	Cum. leachate : 111.9
Cum. evaporation : 14.4	Cum. evaporation : 21.2
Ponded water : 16.1	Ponded water : .0
Cum. transpiration : 129.3	Cum. transpiration : 190.4
Prof. water change: 156.5	Prof. water change: 123.0
Profile water cont.: 218.4	Profile water cont.: 184.9
Crop cover fraction: .9000	Crop cover fraction: .9000
Mass balance error: 1.2	Mass balance error: 1.4

Depth mm	Theta	Potl. kPa	Flux mm	Root distrib
50	.275	-1.4	121.54	.100
150	.275	-1.4	102.96	.200
250	.275	-1.4	79.53	.200
350	.275	-1.4	58.35	.150
450	.271	-1.5	39.69	.100
550	.249	-2.2	22.46	.050
650	.201	-6.1	10.05	.050
750	.132	-43.4	2.02	.050
850	.115	-81.1	.73	.050
950	.115	-81.0	1.14	.050
Drainage flux :			1.81	

Depth mm	Theta	Potl. kPa	Flux mm	Root distrib
50	.168	-14.0	100.19	.100
150	.171	-12.8	106.96	.200
250	.174	-11.8	106.97	.200
350	.178	-10.6	106.51	.150
450	.183	-9.4	107.30	.100
550	.188	-8.3	108.61	.050
650	.192	-7.5	110.17	.050
750	.195	-6.9	106.54	.050
850	.199	-6.4	95.88	.050
950	.201	-6.1	83.84	.050
Drainage flux :			72.36	

(Water fluxes are cumulative since the previous printout and, except for the drainage flux, refer to the upper boundary of each depth segment.

Day 80.	Day 100.
Date 8/ 2/54	Date 8/22/54
Time24h 0 1.0000000Day	Time24h 0 1.0000000Day
Cum. infiltration: 650.7	Cum. infiltration: 723.9
Could not infiltrate: 27.0	Could not infiltrate: 27.0
Cum. leachate : 237.0	Cum. leachate : 280.3
Cum. evaporation : 24.3	Cum. evaporation : 29.3
Ponded water : .0	Ponded water : .0
Cum. transpiration : 218.8	Cum. transpiration : 264.0
Prof. water change: 169.0	Prof. water change: 148.7
Profile water cont.: 230.9	Profile water cont.: 210.6
Crop cover fraction: .9000	Crop cover fraction: .9000
Mass balance error: 1.5	Mass balance error: 1.6

Depth mm	Theta	Potl. kPa	Flux mm	Root distrib	Depth mm	Theta	Potl. kPa	Flux mm	Root distrib
50	.215	-4.4	199.65	.100	50	.191	-7.6	68.18	.100
150	.221	-3.9	189.94	.200	150	.196	-6.8	64.28	.200
250	.226	-3.5	175.29	.200	250	.202	-6.0	55.25	.200
350	.230	-3.2	163.91	.150	350	.207	-5.3	49.08	.150
450	.233	-3.0	155.42	.100	450	.212	-4.7	45.17	.100
550	.235	-2.9	148.51	.050	550	.215	-4.4	43.04	.050
650	.237	-2.8	142.84	.050	650	.218	-4.1	42.67	.050
750	.238	-2.7	137.59	.050	750	.220	-3.9	42.38	.050
850	.238	-2.7	132.89	.050	850	.222	-3.8	42.33	.050
950	.237	-2.8	128.73	.050	950	.223	-3.7	42.63	.050
Drainage flux : 125.11					Drainage flux : 43.27				

(Water fluxes are cumulative since the previous printout and, except for the drainage flux, refer to the upper boundary of each depth segment.

Day 120.	Day 128.
Date 9/11/54	Date 9/19/54
Time24h 0 1.0000000Day	Time24h 0 1.0000000Day
Cum. infiltration: 782.4	Cum. infiltration: 783.5
Could not infiltrate: 27.0	Could not infiltrate: 27.0
Cum. leachate : 328.5	Cum. leachate : 338.7
Cum. evaporation : 31.7	Cum. evaporation : 32.4
Ponded water : .7	Ponded water : .0
Cum. transpiration : 285.5	Cum. transpiration : 292.0
Prof. water change: 135.2	Prof. water change: 118.7
Profile water cont.: 197.1	Profile water cont.: 180.6

Crop cover fraction: .9000
Mass balance error: 1.6

Depth mm	Theta	Potl. kPa	Flux mm	Root distrib
50	.197	-6.7	56.15	.100
150	.191	-7.7	50.36	.200
250	.190	-7.9	42.92	.200
350	.192	-7.5	39.75	.150
450	.195	-7.0	39.17	.100
550	.197	-6.6	39.76	.050
650	.200	-6.2	41.12	.050
750	.202	-5.9	42.72	.050
850	.203	-5.7	44.49	.050
950	.204	-5.6	46.32	.050
Drainage flux :			48.16	

Crop cover fraction: .9000
Mass balance error: 1.6

Depth mm	Theta	Potl. kPa	Flux mm	Root distrib
50	.169	-13.7	.34	.100
150	.171	-12.9	1.68	.200
250	.173	-12.0	1.50	.200
350	.177	-11.1	1.59	.150
450	.180	-10.1	2.22	.100
550	.183	-9.3	3.29	.050
650	.186	-8.7	4.64	.050
750	.188	-8.3	6.04	.050
850	.189	-8.0	7.44	.050
950	.190	-7.8	8.85	.050
Drainage flux :			10.27	

(Water fluxes are cumulative since the previous printout and, except for the drainage flux, refer to the upper boundary of each depth segment.