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GENERATION OF LEACHATE AND THE FLOW REGIME IN LANDFILLS

by

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Abstract

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In this thesis the generation of leachate and the presence and movement of water in landfilled municipal solid waste (MSW) is investigated.

The precipitation-leachate discharge relationship for landfills was found to be dominated by evaporation, accumulation in the soil cover, accumulation in the solid waste and fast gravitational flow in a network of channels.

The flow regime is governed by the heterogeneity of the internal geometry of the landfill, which is characterized by a discrete structure, significant horizontal stratification (resulting from the disposal procedure), structural voids, impermeable surfaces, and low capillarity. Also the boundary conditions, that is the water input pattern, has shown to be important for the flow process. Based on this, landfilled waste can be conceptualized as a dual domain medium, consisting of a channel domain and a matrix domain. The matrix flow is slow and diffusive, whereas the channel flow is assumed to be driven solely by gravity and to take place as a thin viscous film on solid surfaces.

A kinematic wave model for unsaturated infiltration and internal drainage in the channel domain is presented. The model employs a two-parameter power expression as macroscopic flux law. Solutions were derived for the cases when water enters the channel domain laterally and when water enters from the upper end. The model parameters were determined and interpreted in terms of the internal geometry of the waste medium by fitting the model to one set of infiltration and drainage data derived from a large scale laboratory experiment under transient conditions. The model was validated using another set of data from a sequence of water input events and was shown to perform accurately.

A solute transport model was developed by coupling a simple piston flux expression and a mobile-immobile conceptualization of the transport domains with the water flow model. Breakthrough curves derived from steady and transient tracer experiments where interpreted with the model. The transport process was found to be dependent on the boundary conditions.

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CONTENTS

This thesis is based on the following papers, which are referred to in the text by their Roman Numerals:

I Bengtsson, L., Bendz, D., Hogland, W., Rosqvist, H., Åkesson, M (1994) Water balance for landfills of different age, J. Hydrol., 158:203-217.

II Bendz, D., Bengtsson, L. (1996) Evaporation from an active uncovered landfill, J. Hydrol., 182:143-155.

III Bendz, D., Singh, V.P., Åkesson, M. (1997) The accumulation of water and the generation of leachate in a young landfill., J. Hydrol., 203:1-10.

IV Bendz, D., Singh, V.P., Bengtsson, L. (1997) Hydrological characteristics of landfills - implications for modeling, (Submitted)

V Bendz, D., Singh, V.P., Rosqvist, H., Bengtsson, L. (1997) Kinematic wave model for water movement in municipal solid waste, Water Resour. Res. (In Press)

VI Bendz, D., Singh, V.P. (1997) Solute Transport under Steady and Transient Conditions in Municipal Solid Waste. (Submitted)

INTRODUCTION

Management of municipal solid waste

With industrialization and as the worlds population have become increasingly urbanized the production of municipal solid waste (MSW) has increased. The production of municipal solid waste in the industrialized countries varies from under 1 kg per person per year, e.g. Germany, to 2 kg per person per year in the USA according to figures given by the US Environmental Protection Agency [EPA, 1997].

MSW may be landfilled, incinerated, composted or recycled. Incineration and landfilling are the dominating waste management methods. Today the proportions of MSW which are landfilled and incinerated vary greatly between industrialized countries [Carra and Cossu, 1990]. Switzerland, where 80% of the MSW is incinerated and 20% is landfilled, represents one extreme. Canada and Finland represent the other extreme, where 95% of the MSW is landfilled. The UK and the USA can also be found at this end of the scale, where 88% and 83% of the MSW is landfilled, respectively. In Sweden, where this study has been carried out, approximately 35 percent is landfilled and 60 percent is incinerated. However, the trend in the developed countries is for waste management to become better economically and environmentally optimized by dividing the MSW stream in different categories using the appropriate technology. This is called "integrated solid waste management" and is based on a hierarchy of waste managing methods. They are listed here according to their degree of priority:

1. Waste reduction
2. Recycling
3. Composting
4. Incineration
5. Landfilling

As can be seen, landfilling represents the last option. Reducing the amount of waste produced and the degree of recovery/recycling have the highest priority. Waste that is not recyclable may contain energy or nutrients which can be recovered by composting or, as the second alternative, by incineration. By incinerating, the waste becomes stable and a considerable decrease in volume is gained. An effect of integrated waste management is that the volume of waste which is landfilled will decrease in the future. The composition will change; the organic fraction of the landfilled waste will be significantly reduced, and hazardous fractions such as, incineration residues, will increase [Christensen et al., 1992].

With a growing environmental concern in the sixties and seventies, landfills were acknowledged as a major threat to groundwater and surrounding surface waters. The landfills at the time were more or less just waste dumps, providing a cheap method of waste disposal, and little effort was made to collect leachate and biogas. Starting as a local problem the landfilling became a national and international issue as a result of the legislation introduced in many countries when environmental problems related to landfills were acknowledged. In Sweden, landfilling was formally defined as an environmental problem in the environmental protection act of 1969 and it was ruled that a permission was required to construct and manage a landfill. This resulted in a concentration of landfilling to fewer and larger sites operated in sanitary manner. This is a trend that Sweden shares with all other industrialized countries.

Different landfill strategies have been adopted in different countries depending on the role the landfill is expected to play. In the USA and Germany, the containment or encapsulation philosophy prevails [Carra and Cossu, 1990]. This implies a landfill design which is focused on isolating the landfill from the surrounding environment. The design features top and bottom lining and a leachate collection system. The leachate is processed in a waste-water treatment plant. Other countries such as Canada, Denmark and Sweden have not adopted the containment strategy. Instead, the landfill is regarded as a waste stabilization plant, where the energy and mass gradients between the landfill and the environment are equilibrated in a controlled manner to a "final storage quality", when the emissions are considered not to significant affect the existing concentrations in the soil, air and water [Brunner and Baccini, 1989]. The landfill can thereby become an integrated part of the environment. The dilution and attenuation strategy is strongly embraced in the UK. The landfill is regarded as plant which transfers matter to the lithosphere at a rate and to an extent that the environment can handle. The philosophy relies on the potential of the attenuation processes which reduce the contaminant concentration during transport through the underlaying strata.

Co-disposal of MSW, industrial, and hazardous waste is also a controversial matter. This method of waste disposal has been banned in many countries, such as Germany and USA, due to bad experiences from old-fashioned landfills and dumps [Christensen et al., 1992; Watson-Craik and Sinclair, 1995]. The idea behind co-disposal of MSW and hazardous waste is to utilize the inherent capacity of a MSW landfill to attenuate and assimilate pollutants or, in other words, to exploit the landfills as anaerobic filters. Co-disposal is a well accepted practice in the UK, given that the amount of disposed hazardous waste is kept at a level where it does not negatively affect the degradation process and leachate and gas quality [Cossu, 1990].

The European Commission adopted a proposal for a Council Directive on the landfilling of waste in march 1997 [Petersen, 1997]. The objective of the proposal is to establish high standards for the management of waste and to encourage waste reduction through recycling and reuse. The integrated waste management principle is emphasized. The requirement of waste pretreatment prior to landfilling is included in the proposal. The requirements for water control and leachate management include measures to be taken to prevent the infiltration of precipitation and inflow of surface- and groundwater. Plastic bottom lining will be compulsory for all landfills. For hazardous landfills, a top liner will also be required. Leachate is to be collected and treated. Greenhouse gas emissions have been given the highest priority and a reduction in the landfilling of waste is regarded to be the most cost-effective method of reducing such emissions. If organic waste is landfilled, biogas generation cannot be avoided. With the aim of reducing the emission of methane a limit is proposed for the landfilling of biodegradable waste, and provisions made to ensure that biogas from new as well as old landfills is collected. By the year 2010, the biodegradable fraction of waste which is landfilled must be reduced by 75% (by weight) compared with the amount of landfilled biodegradable waste in 1993. The idea is to reach this goal by promoting separate waste collection, sorting, composting and recycling.

Joint disposal of hazardous and non-hazardous waste will be banned. Further, the price currently charged for landfilling is not regarded to reflect the true cost of waste management by landfilling to society and environment. It is believed that increasing landfilling charges will stimulate a reduction of the generated waste volume.

MSW is a broad and ill-defined term, covering all the waste generated by households and light commercial activities. Local conditions in the community may also have a major impact on the composition of the MSW produced. According to a compilation of data from a number of industrialized countries by Carra and Cossu [1990], MSW is typically made up of about 30% organic material, 34% paper, and 6-8% each of glass, plastic and metals (the figures are medians of data from 14 industrialized countries). MSW in developing countries typically has a much larger fraction of organic material, about 60-75% [Diaz et al., 1996]. The percentage of plastics shows an increasing trend all over the world.

Environmental Concern

There are gradients of matter and energy between the landfill and the surrounding environments. By simply referring to the second law of thermodynamics, of spontaneous increase in entropy, it can be stated that, with time, the energy level in a landfill will approach the level of the surroundings. This means that in the long term, matter and heat will leave the landfill unless the storage of energy and matter is maintained by a continuous input of energy. Since landfilling is a relatively new waste management practice, no long-term records of the mass flow out of a landfill are available. It can therefore only be speculated how long it may take before equilibrium is reached, that is when the energy level in the landfill is equal to that of the surrounding environment.

A landfill containing biodegradable waste can be regarded as a biological reactor consisting of three phases: solid waste, liquid (leachate), and gas (mainly CO_2 and CH_4) [Christensen, 1992]. The presence of water is important since it is necessary to redistribute chemicals, microorganisms and nutrients within the landfill [see, e.g Augenstein and Pacey, 1991; Christensen and Kjeldsen, 1989]. Matter may be transported within the water by molecular diffusion and with the water medium itself by convection. Both a high water content [Ehrig, 1991] and flux [Klink and Ham, 1982] have been found to enhances the biochemical processes. Before the microorganisms can start to degrade organic matter it must be split through hydrolysis and solubilized. The hydrolysis may be rate limiting for the whole degradation process [Leuschner and Melden, 1983].

From the time of disposal, the waste goes through a number of degradation phases which govern the time pattern of the rate and composition of the emissions. An idealized anaerobic waste degradation sequence as proposed by Christensen and Kjeldsen [1989], based on the work of Farquhar and Rovers [1973] and Ehrig [1987], includes five phases.

I. After landfilling, aerobic conditions prevails for a short period of time. During this phase, easily degradable organic material is converted into water and carbon dioxide under the consumption of oxygen and nitrogen.

II. When the oxygen that was entrapped in the waste during the landfilling procedure has been exhausted, facultative and anaerobic bacteria become active. The decomposition products are mainly volatile fatty acids (VFA) and carbon dioxide. The organic acids reduce the pH and lead to a high chemical oxygen demand, COD, in the leachate. The acid environment is toxic to methane-producing bacteria and methane production does not take place.

III. As methanogenic bacteria become more dominant, the VFA are converted into methane and carbon dioxide. As a result, the pH increases and the COD declines.
IV. This is the stable methane-producing phase with a methane concentration in the gas of about 50-65% by volume, the rest is mainly carbon dioxide.
V. Only the organic matter which is most difficult to degrade remains to be converted, the degradation process slows down and the methane production rate declines.

Leachate consists of water and a cocktail of soluble organic, inorganic, and bacterial compounds together with suspended solids. The components of leachate can be divided into four groups of pollutants. Organic matter, expressed as COD or total organic carbon, TOC, specific organic compounds, inorganic compounds, and heavy metals [Christensen, 1992]. The inorganic components are positive ions such as ammonium, iron, magnesium, manganese, calcium, sodium, potassium, and negative ions such as chloride and hydrogen carbonate. Compared with sewage water, leachate is typically composed of a high concentration of organic compounds, a higher concentration of nitrogen and a lower concentration of phosphor. High concentrations of heavy metals are seldom found in the leachate due to that stable sulfide-metal complex is formed in the anaerobic environment of the landfill interior [Christensen and Kjeldsen, 1995; Flyhammar, 1997]. The composition of the leachate is site specific and depends on the type of waste, the rate of decomposition, and the presence and mobility of water [Ehrig, 1983; Britz, 1995]. Leachate management (collection and treatment) is a costly procedure and is complicated by the composition and strength of the leachate which, in addition to long-term trends, may show a daily and seasonally variability [Britz, 1995].

The leachate will constitute an environmental problem for hundreds of years to come [Christensen et al., 1992]. Despite the fact that groundwater contamination constitutes a serious threat to the environment, the impact of leachate is local and is limited to a range of about 500 meters [Christensen, 1992, Gustavsson and Holm, 1989]. This is due to a variety of retention mechanisms through which the chemicals that escape the landfill with the leachate through the lower boundary will undergo as they travel through the soil. These mechanisms include: mechanical filtration, restriction of the flow of suspended contaminants, precipitation and co-precipitation, adsorption, dilution and dispersion, microbiological activity, and volatilization [see for example, Christensen, 1992].

Globally, there are large variations in the generation and emission of methane from landfills. Landfill gas control started in the USA in the late 60s and early 70s, was introduced in Germany in the mid 1970s, and has now been implemented in most developed countries [Christensen et al., 1996]. The gas is either utilized or flared. However, in a global perspective, the sites where gas collection systems have been installed are in minority. In the USA and Western Europe, only about 30 % of the produced landfill methane is collected, whereas in Africa, Central and South America, and Asia the waste ends up in uncontrolled waste dumps where no methane whatsoever is collected [Meadows et al., 1997]. Due to urbanization and increasing population in developing countries, methane emission is likely to increase significantly in the future. The trend in the developed countries will, however, be the opposite. This will be the result of increasing incineration, composting and recycling which will reduce the amount of biodegradable waste being landfilled [Meadows et al., 1997].

Even if a landfill is equipped with a gas collection system, gas will escape, directly from the uncovered waste, through the top cover and by migration through the surrounding soil [Kjeldsen, 1996]. Different scales of impact from biogas emission were identified by Luning and Tent [1993]. On the local scale uncontrolled methane gas may migrate to nearby structures leading to the risk of explosion. Methane may also kill vegetation which, in turn, can cause erosion problems. At the local scale a landfill can also give rise to odour nuisance. However, the most severe environmental impact from biogas emission is on the global scale. Methane is a greenhouse gas with a large global warming potential, 21 times that of carbon dioxide. Biogas also contains some tracer gases, such as CFCs (chlorofluorocarbons) which are very aggressive greenhouse gases which also destroy the ozone layer. According to the International Panel of Climate Changes (IPCC), landfills contribute with about 8-20 % of the anthropogenic methane emission [Thorneloe, 1996].

From this introduction it can be concluded that, until the point is reached where only recyclable or reusable products are designed and manufactured, landfills will be needed. Landfills are a basic resource in the integrated solid waste management system which is implemented in most developed countries. In addition, landfilling may constitute the main treatment method in developing countries, due to its flexibility and relatively simple technology [Diaz et al., 1997], and in sparsely populated areas in developed countries. It may also be used as back-up in case of a shutdown of the ordinary incineration plant.

The aim in modern landfill management is to control the process and to minimize the environmental impact by equilibrating the mass and energy gradients between the landfill and the surrounding environment in a controlled manner. This requires knowledge and understanding of the biogeochemical processes that govern the production of biogas and leachate. Regardless of the landfill philosophy employed, water plays a key role and several researchers have emphasized the need for further investigations concerning the presence and mobility of water and the transport of pollutants [Straub and Lynch, 1982; Augenstein and Pacey, 1991].

SCOPE AND AIMS

The aim of this study was to investigate the mechanisms that govern leachate generation in landfills and, based on the hydrological characteristics, design a model for flow and transport.

Only municipal solid waste landfills (MSW) constructed in a conventional way, with a soil cover that permits a certain degree of infiltration are considered here. Each landfill is a unique system. Therefore, this study focuses on highlighting the characteristics and features of the landfill formation, rather than providing exact figures or parameter values for certain processes. A framework is developed for the analysis of the landfill as a hydrological system and for modeling flow and conservative transport processes in the landfill interior with a continuum approach.

THE LANDFILL AS A HYDROLOGICAL SYSTEM

A landfill is a highly heterogeneous and anisotropic formation with a complex internal geometry that is variable in time due to biodegradation and subsequent settling [Paper IV]. Water may enter the landfill as precipitation, with the waste itself as the initial water content or, if the landfill is not properly sited, as surface- or groundwater flowing in. Water may also be formed by the anaerobic degradation process. However, this volume is estimated to be small and is therefore neglected [Paper I]. For properly sited landfills, the atmospheric fluxes, precipitation and evaporation, and overland flow will govern the input to the landfill system. In the long-term perspective, all water that infiltrates a landfill and percolates down below the root zone will form leachate, [Paper I, Stegmann and Ehrig, 1989]. In a young landfill, however, water is accumulating, and only a small part of the infiltrating water forms leachate. As the landfill becomes older, and the water storage potential is exhausted, an increasing portion of the infiltrating water becomes leachate until steady-state conditions are reached, i.e. when all water that percolates through the soil cover becomes leachate.

Starting with a systems approach, the external fluxes and the operation of the system were investigated, but not the system itself nor the governing physical laws [Paper I,II,III]. The intention of a systems approach is to avoid the difficulties that arise from the complexity of the system structure and the complexity of the governing physics. The system structure and the governing physics are joined together to form a single concept called the system operation and constitutes the operation performed by the system on the input and the throughput in order to transfer input into output. Thus, the system operation is the link between input and output. A system can be composed of a number of subsystems, each of which can have a distinct input-output linkage. The various components of a hydrological system can be regarded as hydrological subsystems. A system may be: linear or non-linear, time variant or time invariant, lumped or distributed, deterministic or stochastic. A system is linear if it satisfies the property of proportionality and the property of superposition, is time variant if the input-output relationship changes with time, lumped if the spatial variability is ignored, deterministic if the system uniquely defines its output for a specified input and initial and boundary conditions, and is stochastic if its behavior is governed by laws of probability, e.g. there is a certain probability that a certain output will be obtained for a specified input.

A spatially lumped hydrologic system can be described by the continuity equation

$$I - Q = \frac{dS}{dt} \qquad (1)$$

where I is the rate of water input, Q is the runoff flux, and S is the storage. If the relation $S = f(Q)$ is known, S can be eliminated and (1) can be solved for Q. By treating landfills as hydrological systems the variables Q, I, and S and their variation with time were investigated in Papers I, II, and III, respectively.

The water budgets for landfills of different ages at Spillepeng disposal site in the city of Malmö, Sweden, were studied in Paper I. These are an old closed landfill, an active biocell and six pilot scale landfills. The old landfill and the pilot scale landfills are covered with a one meter thick vegetated soil cover, whereas the biocell is uncovered.

Features of the precipitation-leachate production relationship, such as the importance of water accumulation dynamics and presence of fast channel flow were identified. In addition to the accumulation of water in the waste [Paper III] the other abstractions in the precipitation - leachate production relationship are surface runoff, storage in the soil cover and evaporation [Paper I]. The overland flow has been shown to be of minor importance in several investigations due to the development of cracks, even if low-permeability soils are used [Paper III, Booth and Price, 1989; Ettala, 1987; Ham and Bookter, 1982; Karlqvist, 1987; Nyhan et al., 1990].

Water Input

Given a certain precipitation, the input to the waste domain is governed by the soil cover. The function of the soil cover includes smoothening of the time distribution of the rainfall, due to hydraulic resistance and storage capacity of the soil cover, and water losses due to evapotranspiration which extracts water from the soil. By measuring the soil water content in the grass vegetated soil cover at the pilot-scale landfills it was found that the soil water content in the one-meter thick soil cover showed large fluctuations at small depths [Paper III]. The soil water content was steadily high throughout the year at deeper levels, showing only minor fluctuations. This is in agreement with observations made by Booth and Price [1989].

For a landfill with a well-vegetated soil cover the actual evaporation can be assumed to be comparable to the regional evaporation. For an uncovered landfill, during its operational phase, the conditions are different. Several researchers, for example Christensen and Kjeldsen [1989], have reported aerobic conditions in the top layer of uncovered landfills. Although the net radiation dominates the energy budget at the landfill surface, the heat generated by aerobic biodegradation gives a significant contribution to the net energy supplied to the surface. Since the change in heat storage is small, the net energy is mainly divided into sensible heat and latent heat. Further, the increased surface temperature will induce a vertical density gradient which will increase the vertical mass and heat transport due to the effect of buoyancy. In the investigation of an uncovered biocell at the Spillepeng disposal site during its operational phase, reported in Paper II, the temperature gradient in the top layer of the biocell was measured. The upward heat flux originating from aerobic biodegradation was calculated and was found to enhance the net energy budget by 20%. Since the heat flux was found to be almost constant in time its relative importance was increased during fall and winter when the net radiation decreases. By employing the similarity theory of Monin and Obukhov [1954], which expresses the buoyancy effect as a dimensionless variable, the Obukhov length, the sensible heat flux was calculated. The latent heat flux was then determined as the rest in the surface heat budget equation. The biological heat was found to enhance the actual evaporation by about 10%.

There are two other processes which might offset each other. The coarse structure of the compacted waste facilitates rapidly channeling of rain water to greater depths, while the upward temperature gradient creates a vertical transport of moisture from lower levels to the surface where it can evaporate. The relative importance of these processes for the actual evaporation was not investigated.

Storage and Leachate Flux

According to a summary of the literature given in Paper I the initial volumetric water content of MSW is about 0.15-0.20 and the field capacity is about 0.40. The field capacity is defined as the maximum water content that the capillary forces in a porous medium can retain against gravity. Thus, the volumetric water content may increase by approximately 0.2-0.25 before the field capacity is reached.

Theoretically, water cannot move downwards until the entire landfill has reached field capacity. Nonetheless, due to the internal geometry and heterogeneity of the landfill formation, water is flowing in restricted channels or in local regions were field capacity has been reached [Paper I, IV]. The phenomenon is well documented in the literature, see for example, Blakey [1982], Blight et al. [1992], Burrows et al. [1997], Harris [1979], Ham and Bookter [1982], and Zeiss and Major [1993]. The rate of accumulation is dependent upon the presence of channel flow and factors such as waste composition and age, initial moisture content, density, and internal geometry [Paper IV, Blakey, 1982; Holmes, 1983].

In Paper I it was found that the accumulation of water still dominated the precipitation-leachate production relationship 10 years after a landfill was closed. Only a weak seasonal dependence of the leachate flux was observed, whereas the leachate production from the pilot scale landfills showed large fluctuations, which were attributed to a continuous system of channels. The storage-leachate flux relationship exhibited by pilot-scale landfills at the Spillepeng disposal site was investigated during the first seven years after the disposal of the waste [Paper III]. The landfills were regarded as a time-variant, deterministic systems, represented by two subsystems; the soil cover and the compacted waste formation. Each subsystem was treated as a black box. The output from the cover subsystem constituted the input to the waste domain and was calculated using the continuity equation. The change in storage in the soil cover was measured during one year on regular basis. Overland flow was also measured but was found to be negligible and was not considered. The annual evapotranspiration from the landfills was assumed to be comparable with the regional evapotranspiration represented by the nearby small watershed Höje river and was determined by a simple water budget, neglecting deeper groundwater recharge. The evapotranspiration on a monthly basis was calculated by using Penman's definition as an index indicating the fraction of the annual actual evapotranspiration that could be attributed a certain month.

It was found that the water storage showed a seasonal dependence. The proportion of the infiltrating water that was discharged as leachate increased from zero to about half during a period of 6.5 years. This corresponds to a total accumulation of 0.18 (m^3/m^3) of landfilled waste. The storage-leachate flux relationship was quantified and was found to consist of a family of curves depending on the storage and the rise or decline of leachate flux.

FLOW AND TRANSPORT PROCESSES

Following the systems approach a closer look into the system, or black box, was taken. Some important hydrological characteristics of landfills were given in the introductory sections of Papers I, II, and III. By discussing important characteristics of landfilled waste, such as channel flow, internal geometry, spatial and temporal variability and their implications for modeling, the foundation was laid for a continuum approach in Paper IV.

Process Assumptions

Due to the compaction procedure, the internal geometry of landfills is highly stratified and exhibits anisotropy in the vertical plane. Based on field observations, the landfilled waste medium may be physically conceptualized as discretely hierarchical, and composed of porous lens-shaped elements with a partially impermeable surface. These lenses represent compacted refuse bags or sacks. For modeling purposes the landfilled waste may be regarded as a dual-domain medium spatially variable at three scales and temporally variable [Paper IV]. The different scales are the following:

(1) The lens scale, which is of the order 0.5 m in the horizontal plane and 0.1 m in the vertical plane.

(2) The truck-load scale, characterizing the length scale of a group of lenses *in situ*, originating from the same garbage truck load (they can be assumed to share certain characteristics), which is of the order of 10 m in the horizontal plane and 1 m in the vertical plane.

(3) The landfill scale, which is of the order 100 m in the horizontal plane and 10 m in the vertical plane.

The geometry and the heterogeneity scales of two compacted MSW layers, separated by a daily cover, are illustrated in figure 1.

The structural voids between the lenses form a network of flow paths or channels in which the water is assumed to move as a thin viscous film, driven by gravity on the solid surfaces that constitute the boundaries of the channel. Capillary potentials have been found to be small in landfilled waste, particularly in fresh waste due to its course structure. The capillary forces were therefore assumed to only play a static role in the channels, that is, their effect may be taken into account as a loss term in the channel flow equations.

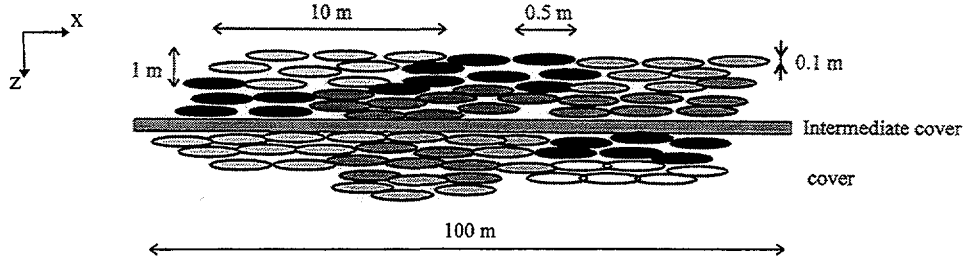

Fig.1 Geometrical configuration of two layers, separated by a daily cover, in an MSW landfill. The figure shows refuse lenses (compacted and partially torn refuse bags), originating from different garbage truck loads, separated by structural voids.

The channels constitute only a fraction of the landfill and act as shortcuts through the bulk of the waste volume, the matrix. The capillary forces play a significant role in the matrix domain where the flow is assumed to be slow and diffusive. The theory for unsaturated flow in porous media can thus be applied here. The interaction of the domains can be described by exchange terms.

Treating the landfill as a discrete flow path medium instead of as an equivalent porous medium can be justified by the observed phenomenon of channeling and by assuming that the intersection points of the flow paths are few on the particular scale of interest, which is smaller than a representative elementary volume (REV).

Breakthrough curves obtained from conservative tracer experiments on laboratory and pilot scale show an early steep rise and an extensive tailing [Paper VI; Rosqvist and Bendz, 1998, Rosqvist et al., 1997]. This behavior was assumed to be the effect of a small solute transport volume and diffusional exchange between the channel domain and a large stagnant water volume in the matrix. This is the mobile-immobile conceptualization originally proposed by Coats and Smith [1964] and further developed by Van Genuchten and Wierenga [1976]. In Paper VI the immobile domain is divided into two sub-immobile domains. The region closest to the channels which constitute the boundaries is the active immobile domain. This domain is assumed to show a fast response to changes in the mobile concentration in the channel. Beyond this is the passive immobile domain which is passive in the sense that it responds slowly to concentration changes in the mobile domain. This three-domain approach is justified by the small ratio between the solute transport volume and the total volume. The diffusional transport between the domain was assumed to be governed by a first-order mass transfer expression.

Governing Equations

The flux laws employed in hydrology are commonly either convective or diffusive. A general flux law may be written as [Singh and Prasana, 1998]

$$u = \alpha h^m + \beta(h)\frac{\partial h}{\partial x} \tag{2}$$

where α, β, and m are parameters, h is the concentration, such as water depth in the case of overland flow, water content in the case of subsurface flow and solute concentration in the case of a transport process. The equation is composed of two parts, the first term on the right-hand side is the convective part and the second term is the diffusive part. The governing equations employed in hydrology for flow and transport processes constitute special cases of equation (2). By combining equation (2) with the continuity equation a general flux equation is obtained:

$$\frac{\partial h}{\partial t} + c\frac{\partial h}{\partial x} - \frac{\partial}{\partial x}\left(\beta(h)\frac{\partial h}{\partial x}\right) = 0 \tag{3}$$

where

$$c = m\alpha h^{m-1} \tag{4}$$

Equation (3) is a nonlinear parabolic equation and it can be recognized that Richards equation for unsaturated flow in porous media constitute a special case. Richards equation is obtained by modifying Darcy's law to handle unsaturated flow by incorporating an expression for the relation between the hydraulic conductivity and the water content. Usually the function for unsaturated hydraulic conductivity developed by Brooks and Corey [1964] is used. Richards equation relies on certain assumptions, such as that the movement of water presupposes a local equilibrium between water content and capillary potential. This makes the use of the Richards equation questionable in the case of macropore or channel flow since the water travels faster than changes take place in capillary potential [Germann, 1990]. The reported models for flow and transport in MSW [Ahmed et al., 1992; Demetracopoulus, 1986; Korfiates et al., 1984; Lee et al., 1991; Straub and Lynch, 1982; Vincent et al., 1991] share certain characteristics, such as that they treat MSW as a homogeneous porous medium. Richards equation has been employed to model water flow and the convection-dispersion equation to model the transport process. The assumptions, which made implicitly when applying these models, have not been justified.

If c and β are kept constant equation (3) becomes linear. This is the common convection-dispersion equation (CDE) for solute transport. The CDE relays on the assumption that the dispersion phenomenon can be formulated mathematically as a diffusional flux, although it is a result of convective velocity differences. This assumption can be justified in the case of a homogeneous medium where the transport distance is sufficiently long so that all streamlines have had enough time to mix. For simplicity, and due to the heterogeneity of the MSW medium, the CDE was avoided here. Instead, a simple piston type of flux law, derived by setting $m=1$ and $\beta=0$ in equation (3), was employed [Paper VI].

When outlining a framework for modeling channel flow in landfilled waste Richards equation was not considered appropriate due to its underlying assumptions [Paper IV]. Based on the flow regime and inspired by the work by Germann and co-workers in a series of papers [Beven and Germann, 1981, Germann, 1985, 1990, Germann and Beven, 1985] where the vertical movement of soil moisture in macropores was treated as kinematic waves the following strict convective flux law was employed in this work [Paper IV, V, and VI]:

$$q = b\theta^{a} \tag{5}$$

where a is a dimensionless exponent and b is the channel conductance (ms^{-1}) which can be interpreted as the lumped effect of surface, geometrical, and spatial characteristics of the flow path. The exponent a can be summarized as the impact on the mobile part of the system by the stagnant parts [Germann and DiPietro, 1996]. This flux expression is obtained from (2) by setting $\beta = 0$. Equation (3) now reduces to a nonlinear hyperbolic equation also called kinematic wave equation. Lighthill and Witham [1955] used the term kinematic waves to indicate that the formulation of motion excludes mass and force, in contrast to the dynamic wave. The kinematic wave equation has found wide application within surface-water hydrology and subsurface hydrology (for a complete discussion of the subject see Singh [1996, 1997]). Since the equation is hyperbolic, the method of characteristics can be used to solve the governing equation. A characteristic can be defined as a curve in space and time along which the partial differential equation reduces into a system of ordinary differential equations.

Applicability of the Kinematic Wave Model

The kinematic wave model may be applied to a single member, one channel, of the flow path network. Or, neglecting the spatial variability, the whole network may be lumped into one vertical channel domain. In the former alternative the outflow hydrograph from one channel becomes the inflow, the boundary condition, to the next. Both alternatives are discussed in Paper IV. Regardless of whether the channel domain is lumped or distributed, the water may enter it in two different ways: through the upper boundary or laterally. In Paper IV solutions of the governing kinematic wave equation are derived for these two cases.

To test the performance of the kinematic wave model, water flow and solute experiments were performed for both steady-state and unsteady-state conditions in a large, 3 m^3, undisturbed sample taken from a 22-year-old deposit. In Paper V the model parameters a and b where determined using data from a steady-state experiment. The model was then validated by using data from an unsteady-state experiment. The model was found to perform well but further development to include spatial variability was suggested.

In Paper VI a piston type of flux law was couple to the water flow model presented in Paper V. The solute transport model was used as an interpretation tool and was calibrated to the breakthrough curves derived from both the steady-state and the unsteady-state experiments. By solely adjusting the size of the immobile active domain the model was capable of accurately reproducing the breakthrough curves from both experiments. It was concluded that the solute transport volume in MSW is not only dependent on the structure of the medium but also on the boundary conditions.

CONCLUSIONS

Landfills will continue to be a basic resource in integrated waste management in the developed countries. In a global perspective, it will be the most common waste treatment method in the foreseeable future, due to its flexibility and simple technology. The environmental impact of landfills is mainly associated with the emission of leachate and gas. The leachate that escapes the landfill is subject to a variety of retention mechanisms and its effects are only local, whereas the emission of methane gas has a global impact since it is a green house gas. The goal of sanitary landfilling is to stabilize the landfill in a efficient and controlled way until it has reached a final storage quality and can become an integrated part of the environment. Both the presence and the flux of water thus play a key role in the stabilization process.

The precipitation-leachate discharge relation for a landfill is dominated by evaporation, accumulation in the soil cover, accumulation in the waste and the non-uniform flow field. Water is only flowing in a small fraction of the bulk of waste. Channel flow was identified as dominating process for the movement of water and solutes in landfilled waste and is governed by the internal structure and the boundary conditions, i.e. the water input pattern. The transport process is nonideal since access to large portions of the landfilled waste is restricted by diffusional mass transfer.

It can be concluded that the internal geometry of the waste medium and the non-uniform flow field calls for an appropriate flow model, the waste medium should not, as is commonly done in the literature, be characterized as a homogeneous soil with special properties. In this study it is proposed that the landfill should be conceptualized as a dual-domain medium. Richards equation may be applicable in the matrix domain where flow is slow and can be assumed to be fairly uniform. The flow in the macropore domain occurs as a thin viscous film held by surface tension on solid surfaces and is mainly gravitational. Based on this, a strictly convective power law expression is employed as the governing flux law which, in combination with the continuity equation, produces a kinematic wave equation. The kinematic wave approach has been tested at large laboratory scale under transient conditions with good result.

A simple piston flux expression was coupled to the kinematic wave model and tested on laboratory scale. The observed dispersion of breakthrough curves derived from tracer experiments was explained solely by the reversible mass transfer between mobile water in the channel domain and a large stagnant water volume in the matrix. It was found that the fraction of the immobile domain that was active in the reversible exchange of solute was dependent on the boundary conditions.

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Paper I

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Water balance for landfills of different age

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Abstract

Water-related processes in landfills are discussed with emphasis on internal processes such as field capacity, moisture variation in time and space, and macropore flow. Runoff production and evaporation from landfills in Sweden of different age are investigated. It is clarified in what ways and for how long a closed municipal landfill differs from an ordinary land area from a hydrological point of view.

1. Introduction

The main environmental problem of sanitary landfills is the potential risk of groundwater pollution and subsequent influence on surface water quality. The total pollutant load to the environment is dependent on the quantity and the quality of the water that percolates through the landfill and reaches the groundwater. The subsurface runoff water, i.e. the drainage water, from a landfill is called leachate, and subsequently the quality of this water, leachate quality. The quality of the leachate depends on the initial waste composition of the landfill and on the biological, chemical and hydraulic state of the landfill. As long as a landfill is used, and for some years after it has been closed, the site can be superintended and the leachate collected and treated. In a longer perspective, the site must be regarded as a part of the environment and the emissions should be harmless to the environment. To design landfills and manage them so that the emissions become small, it is necessary to determine the quantity and quality of the leachate. The first step is, therefore, to determine the water balance of a landfill. This depends on meteorological conditions (intensity and distribution of precipitation and potential evaporation), on the hydrau-

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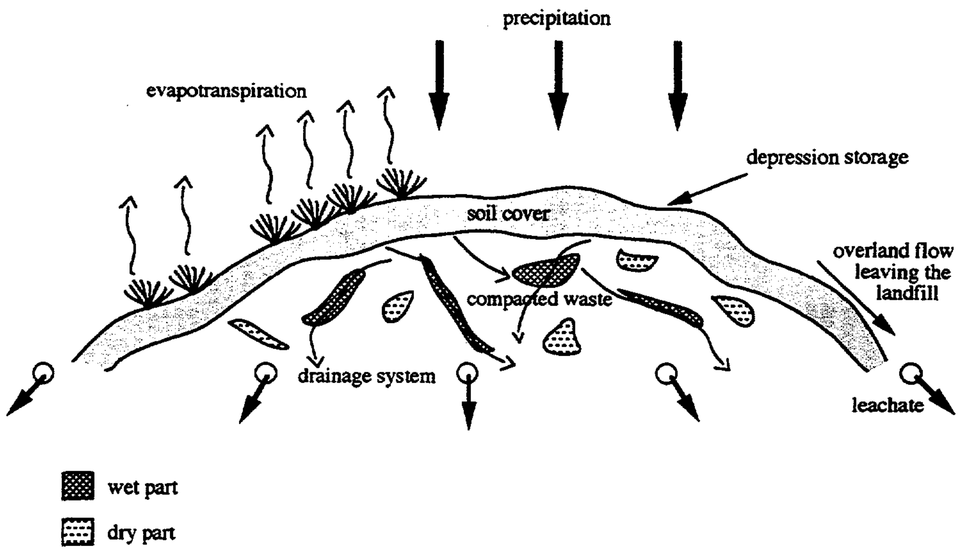

Fig. 1. Water fluxes in a landfill with cover.

lic characteristics and the initial conditions of the waste material, and on biological processes within the landfill. The conditions within the landfill change with time; the character of the biological processes and the hydraulic characteristics change, resulting in changing water balance conditions with time. The way the landfill is managed also influences the water balance, e.g. leachate can be recirculated, waste water treatment sludge can be added, and the landfill can have a more or less permeable surface cover.

In the present paper, first external fluxes from landfills in the form of evapotranspiration and overland flow are discussed. Thereafter, internal water-related processes are dealt with, emphasizing the water storage capacity of the waste material relative to its initial moisture content. With this information, data on leachate production presented in the literature are analysed with respect to seasonal and long-term distribution. Having the theoretical background, the water balance for landfills of different age in the city of Malmö in the southern part of Sweden is quantified on a monthly basis. The storage of water in the landfills is calculated and its influence on the distribution of water discharge from the landfills is shown. Drainage volumes and evapotranspiration from the various landfills are compared, and also compared with the runoff from small agricultural basins.

2. Water fluxes in a landfill

Water is introduced into a landfill through the moisture of the deposited waste, in some landfills with waste water treatment sludge, and as precipitation. Some of the precipitation may run off as overland flow, and some may evaporate from the waste material or be removed by transpiration from a vegetation cover. Some water may be consumed by biological processes. The remainder must accumulate or be discharged by drainage. The water fluxes in a landfill with cover are shown in Fig 1.

To minimize emissions to the groundwater, measures are taken to keep the volume of drainage water low. The waste deposits are covered by soil through which, to obtain the goal of minimizing the drainage volumes, as little precipitation water as possible should percolate into the waste deposits. A vegetative cover is usually established to favour evapotranspiration. When rain falls in seasons of the year with little potential evaporation, infiltrating water still percolates into the waste material. Therefore, to prevent this percolation and to promote overland flow from the landfill, a layer of clay is laid as part of the soil cover. Too low moisture content in waste deposits is, however, disadvantageous in that biological processes are slowed down, and the leachate quality remains poor for a very long time.

The production or drainage water from a landfill is strongly controlled by the soil cover of the landfill. Although rather impermeable soils are used as cover, fractures develop and the overland flow is minor (Ham and Bookter, 1982; Karlqvist, 1987; Booth and Price, 1989; Nyhan et al., 1990). For two landfills in southern Finland, Ettala (1987) showed that the infiltration rate exceeded 60 mm h^{-1} although a clay with very low hydraulic conductivity was used as cover. Booth and Price (1989) attributed the absence of overland flow to the existence of large fracture zones and subsidence depressions. They suggested that there is lateral flow within the soil cover toward permeable zones. Bengtsson et al. (1992) found similar flow conditions for meltwater in frozen agricultural clay soil. When figures on overland flow have been given (e.g. Blakey, 1992), the overland flow has been estimated using runoff coefficients and has not been measured. Using small-scale lysimeters, Ham and Bookter (1992) in the USA found cells covered with clay soils to produce overland flow at the expense of reduced evaporation compared with noncovered cells. From the investigation by Blakey, it is also clear that large overland flow manifests itself in low moisture content in the soil cover and low evaporation rates.

Water is lost from a landfill by evaporation. The evaporation losses may be different from those of a meadow or agricultural field. The potential evaporation may be higher if heat is conducted from the interior of the landfill to the ground surface, but the actual evaporation may be lower if there is no or only a thin soil cover and rather dry waste material below. According to Caffrey and Ham (1974), the high temperature during aerobic decomposition is effective in evaporating moisture. However, when anaerobic conditions prevail, the temperature in a landfill is only of the order of 20°C or less. Still the interior temperature is slightly increased compared with natural soils, and this may enhance the potential evaporation. Temperature measurements in the landfills reported in this paper show that the downward increasing temperature gradient in the cover is less than 1°C m^{-1} in the winter. In the summer the temperature decreases downwards. The temperature in the central parts of old waste deposits is about 15°C. Simple steady-state heat conduction calculations show that the heat conduction to the atmosphere is 1–2 W m^{-2}, which means that the contribution from the increased temperature to increased potential evaporation can only be about 1 mm $month^{-1}$.

The evaporation rate depends on the potential evaporation and the moisture conditions in the surface layer of the landfill. Karlqvist (1987) showed that the moisture in an uncovered landfill in the Malmö region was much too low to maintain evaporation near the potential rate. The annual evaporation was 265 mm as compared with the potential value of 550 mm. With a cover, the evaporation loss was closer to the potential evaporation, but still below it, at 342 mm. The regional basin evaporation is 400–450 mm (Lindh, 1983). For landfills in Germany, Ehrig (1989) gave annual evaporation of 450 mm for newly covered landfills, which is about 100 mm less than the regional basin evaporation.

Blakey (1992) showed in a study near London the importance of vegetation growing on the landfills for evaporation losses. If water balance calculations are done for a London landfill, from which water is directed as overland flow, using data on precipitation, leachate and estimated overland flow given by Blakey, the evaporation is found to be small. However, as the overland flow was only estimated and not measured, it might have been in error.

Soil cover plays an important role in leachate production as discussed above. The thickness and the macro-scale conditions of the cover as well as the moisture status of the waste material seem to be crucial for evaporation and for the possibilities of accumulating water in the landfill and thus for leachate production. Booth and Price (1989) found only small seasonal changes of moisture in the bottom part of a soil cover, which may indicate that, if the soil cover is sufficiently thick, the moisture conditions in the waste are independent of season and that the discharge from a landfill is constant over the year.

3. Hydraulic conditions within a landfill

Moist conditions are required for anaerobic biological processes to be effective. Water is consumed in the microbiological processes within a landfill. Carbon dioxide and methane are the major gaseous products produced in the anaerobic process when acetic acid is broken down. There is water vapour in the biogas emitted from a landfill.

When gas production is measured, water consumption can be calculated as shown for example by Young (1990). The gas production from the landfills in Malmö has been measured. Knowing the gas temperature and assuming moisture saturated gas, the water losses to the atmosphere with the biogas are calculated to be 1 mm year^{-1}. The annual gas production from the test cells, containing 3400 tons of domestic waste and extending over an area of 1600 m^2, is about 68 000 m^3, of which about 50% is methane. From the general law of gases, it is calculated that it requires 0.25 l of water to produce 1 m^3 of methane. Thus, 16 500 l of water is consumed, which means a water loss per unit area corresponding to 10 mm year^{-1}. Thus, in annual or monthly water balances, microbiological processes within a landfill are negligible.

Water is stored in a landfill for shorter or longer periods within waste material, e.g. in package material or in wood, in voids in loose organic material, in macropores or holes between densely packed waste or above impermeable layers, e.g. above large plastic bags. Water does not percolate downwards from a part of a landfill until this part has reached field capacity. Because the field capacity is different for different parts of a landfill, leachate can be generated from a landfill even when the degree of saturation in large parts of the landfill is well below field capacity, as pointed out by, for example, Harris (1979). Holmes (1983) and Blakey (1992) used the term absorptive capacity to denote the condition in a landfill when leachate is just barely produced, but this is not a generally accepted term.

The local field capacity in different parts of a landfill, as well as the absorptive capacity, change with increasing age of the landfill. As long as field capacity is not reached in all parts of a landfill, some of the water percolating into the waste deposits is stored. In some parts of a landfill infiltrating water can accumulate and be stored for years before any water leaves the landfill. In old landfills the effect of the moisture storage is to distribute water infiltrating during rainy months as discharge water over longer periods. Studies by Blakey (1982), Ehrig (1983) and Nyhan et al. (1990) showed that in not very new deposits, the annual storage of water is minor. However, in some of Blakey's test cells water was still accumulating after several years, and even after 18 years Harris (1979) found dry spots in a landfill test cell.

Although in soil science soil moisture is given as volume per unit volume, moisture in landfill waste deposits is often given as volume per unit dry weight. When converting field capacities given for municipal waste as volume per unit dry weight by Quasim and Burchinal (1970), Reinhardt and Ham (1974), Harris (1979), Holmes (1983) and Blakey (1992), the volumetric soil moisture at field capacity is about 0.4. Initial moisture content converted to volumetric soil moisture is in the range 0.15–0.20 when volume per unit dry weight values given by Quasim and Burchinal (1970), Campbell (1983), Holmes (1983) and Farquhar (1989) are used. For test cells, Newton (1977) reported water accumulation corresponding to 0.2 l kg^{-1} dry weight after 2 months, and a final accumulation, at equilibrium between water input and output, of 0.4–0.6 l kg^{-1} dry weight, which is a volumetric moisture increase of about 0.25. The soil moisture increase from initial state until any drainage water leaves a landfill is less, of the order of 0.10 or less (Campbell, 1983; Holmes, 1983; Blakey, 1992). In the southernmost part of Sweden, the annual basin runoff is 200 mm, which, as overland flow is almost non-existent, is the total percolation below the topsoil from which evaporation losses occur. For waste material of initial volume by volume moisture content 0.20 and field capacity 0.40, it takes 1 year for a landfill layer of 1 m to reach field capacity. Thus, for a landfill of 10 m height, it should take 10 years before the leachate production corresponds to the regional basin runoff, provided there is no overland flow and the evaporative losses correspond to the basin value. By comparing the discharge from landfills and small basins it should be possible to estimate when a landfill would reach natural state.

There are also small-scale spatial heterogenities within a landfill because of the presence of large and more or less continuous voids, referred to as macropores, and characterized by higher hydraulic conductivity than the surrounding matrix. Kmet (1982), in discussing the appearance of drainage water before the attainment of field capacity, attributed early leachate production to channelling effects, as also did Blight et al. (1992). It has been found from many studies (Bouma et al., 1980;

Beven and German, 1981), that although macropores make up only a small portion of the total voids, they dominate the unsaturated vertical flow recharging groundwater, especially during periods of large water input, as shown by Wild and Babiker (1976) and by Bengtsson et al. (1992). In the study by Bengtsson et al. it was found that cracks develop in frozen clay soils, but close up as the soil thaws and becomes wetter. Robinson et al. (1987) also found cracks to open, close and change character, and attributed this to drying and wetting of the soil. Thus, the macropore structure and therefore the character of preferential flow change in time and space.

4. Leachate production

There are a number of studies in which landfill discharge has been measured (e.g. Meijer, 1979; Ham and Bookter, 1982; Barber and Maris, 1984; Ehrig, 1989; Blakey, 1992; Blight et al., 1992). Sometimes water balance calculations have been used (Fenn et al., 1975; Kmet, 1982) to determine leachate production. The given volumes of drainage water are difficult to evaluate, as the production rates are for single test cells or landfills and for limited periods of time and are often related to annual precipitation only.

As discussed in the previous sections, the discharge from a landfill depends, apart from rainfall and its distribution in time, primarily on evaporation losses and the redistribution in time of infiltrating water in the soil cover and in the waste deposits. The discharge varies over the year. During periods of high precipitation much water may percolate into the waste deposits below the soil cover. When there is little or no evaporation loss, the discharge increases (e.g. Farquhar, 1989; Nyhan et al., 1990; Blight et al., 1992). For landfills of rather fresh deposits, the seasonal discharge fluctuations seem to be large and the response to rain storms recognizable (Ehrig, 1989). However, the discharge from newly closed landfills is small. Ehrig observed daily peaks of 1 mm day^{-1} for landfills closed less than 1 year before, but there were periods when the discharge almost ceased. The annual volume of drainage water was about 200 mm. The fact that water is drained from a landfill before field capacity is reached, and that the discharge fluctuates and sometimes ceases, indicates that the percolation through the landfill, at least to some extent, is preferential flow and occurs in macropores.

On an annual basis, as long as the moisture content is below field capacity, the annual leachate volume from a landfill is reduced as compared with the volume of percolated water. After some time, of the order of many years, the annual drainage volume corresponds, provided there is no overland flow, to precipitation reduced by evaporation. The discharge from old landfills is rather constant throughout the year (Blakey, 1982; Ehrig, 1989), which means that only little of the drainage water travels in macropores. The annual volumes of drainage water produced from old landfills are larger than those from new ones, as shown by Blakey (1982) and Ehrig (1983). This is in agreement with the previous discussion about moisture storage in landfills. However, as previously pointed out, in some landfills water is still accumulating after 10–20 years.

Summarizing the literature survey and the qualitative theoretical discussion above, it is found that precipitation water is continuously accumulating in waste deposits in landfills for many years. Thus, the leachate production from young landfills is minor. There are macropores in these young landfills, and much of the drainage occurs as preferential flow. In old landfills, the macropores are partly closed or in other ways less effective in transporting percolating water. Instead, water probably percolates through the waste deposits, saturated to near field capacity, as flow through a matrix. The discharge does not show large fluctuations.

5. The Spillepeng landfills in Malmö

The leachate from three landfills at Spillepeng, Malmo was studied on a monthly basis for a 4 year period. These are an old landfill, test cells which were covered 5 years ago, and active biocells. Comparison is made with runoff from two agricultural basins. The landfills are managed by SYSAV, a waste management company owned by nine cities and communities in the very southwestern part of Sweden. SYSAV is responsible for the discharge measurements for the old and the active landfill.

The old landfill was established 40 years ago. The drainage water from a part of the landfill, in which the waste is 3–13 years old, is collected via a drainage system, which drains 50 ha. Parts of the landfill, 15 ha, received municipal waste until 1990. The last part of the old landfill was finally covered in 1991. The height of the landfill is 10–25 m. The whole area is covered with grass and some bushes. The refuse in the old landfill is waste from municipal activities, including households, businesses, shops and light industries. All kinds of refuse were disposed of together.

To study biogas production, landfill leachate quality and their dependence on waste composition, test cells were built, each with an area of 1600 m^2 and of 10 m height at the highest end and sloping at 15% to 2 m height at the other end. They were covered in 1988 with a 1 m depth of clay soil on which grass grows. The test cells are lined by clay and a plastic liner. The leachate from each cell is separately collected via a drainage system. Biogas is withdrawn from the test cells. Two test cells, which contain mixed municipal waste from households and light industry, were used in the present study. A section of a test cell is shown in Fig. 2.

The active landfill, which consists of three biocells, was established in 1990. It covers an area of 5 ha. It is built up in layers of 2 m height. At present, 1993, the height is 8–10 m. The landfill is lined by boulder clay. The leachate is collected in a drainage system at the bottom of the landfill. Biogas is collected from the landfill. The refuse in the biocells is mixed municipal waste, including garden waste, and also waste water treatment sludge with high water content.

The agricultural river basin Höje, the runoff of which is compared with the leachate volumes from the landfills, is 223 km^2. At low flow in the river Höje, the contribution to the flow from the waste water treatment plant in Lund is considerable (Hogland, 1986; Berndtsson, 1990). Therefore, the discharge from the plant has been withdrawn from the measured river discharge when comparing runoff and leachate volumes. The

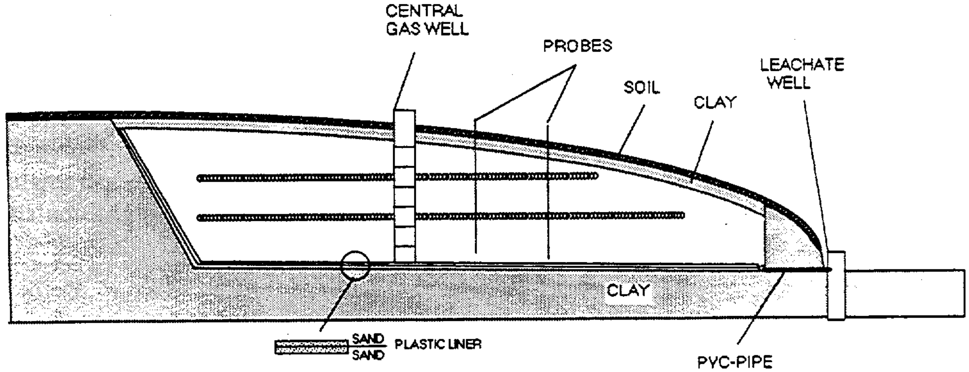

Fig. 2. Section of a test cell.

Höje river basin is flat and well drained. It consists of about 70% arable land. Comparison is also made with the runoff production from the small (about 1.5 km^2) agricultural basin Värpinge (Lindh, 1983).

There is no overland flow from the active landfill nor from the flat old landfill. The test cells slope at 15%, but there are many depressions in the surface cover. As previously discussed and shown, the infiltration capacity for a whole landfill is much higher than the low hydraulic conductivity of the cover material because of fractures in the soil. Summer storms have not been observed to produce overland flow from the test cells, but some minor overland flow has been observed during prolonged rainfall periods in the winter, when the depressions in the cover become full. Still, the annual production of overland flow should be very small.

6. Leachate production from the Spillepeng landfills

The measured monthly leachate production from the old landfill and from the two comparable test cells is shown in Figs. 3–5. Net precipitation, as precipitation minus

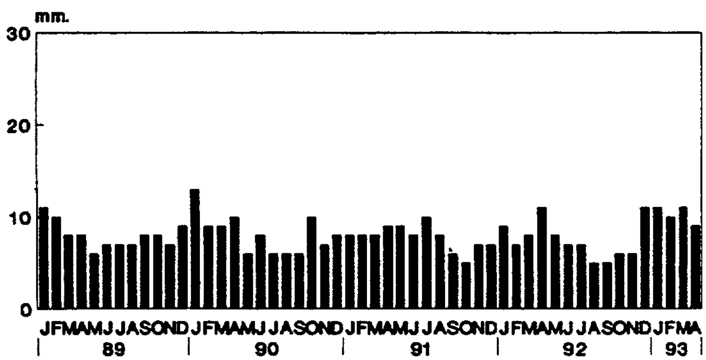

Fig. 3. Leachate production from the old SYSAV landfill at Spillepeng.

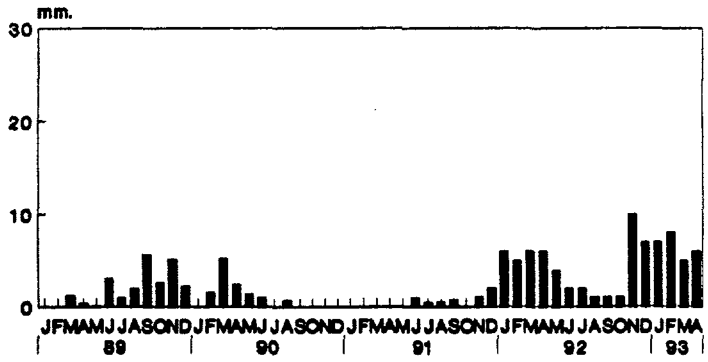

Fig. 4. Leachate production from Test Cell 1, Spillepeng.

potential evaporation, which may be lower than precipitation reduced by actual evaporation, and thus less than the runoff when averaged over time, is given in Fig. 6, to show when leachate could be expected to be produced. The discharge from the old landfill was rather constant, somewhat less than 10 mm month^{-1}, with less leachate in the autumn (5–6 mm month^{-1}) than in the winter and the spring (9–13 mm month^{-1}). The discharge from the test cells was very low (0–2 mm month^{-1}), except for early 1990 and early 1992, when the production was about 5–8 mm month^{-1} for 4-month periods. As a consequence of high precipitation (130 mm) in November 1992, 17 mm of leachate was produced from one of the test cells in November and 20 mm left the cell in December 1992. The drainage rate remained high, and in March 1993 was still 11 mm. The annual volume of drainage water from the old landfill was in the range 80–96 mm and from the two test cells 5–51 mm and 15–82 mm, respectively. These volumes are less by 100–200 mm than values reported by Ehrig (1983) for a region in Germany with 100 mm higher precipitation than in Malmö.

The leachate from the active landfill has, since the measurements started in 1991, excluding the very first month when the production was 44 mm, been in the range 13–27 mm month^{-1}. More water is added to the landfill from waste water treatment

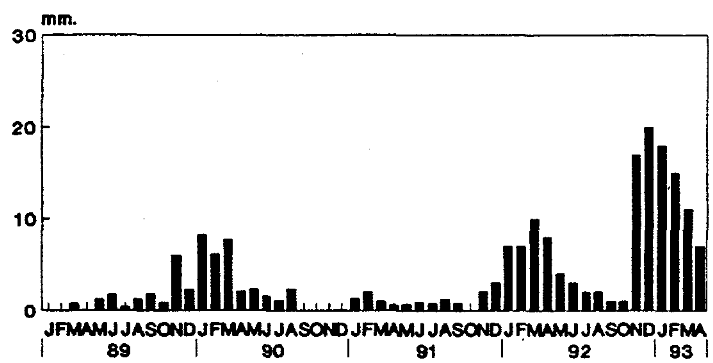

Fig. 5. Leachate production from Test Cell 2, Spillepeng.

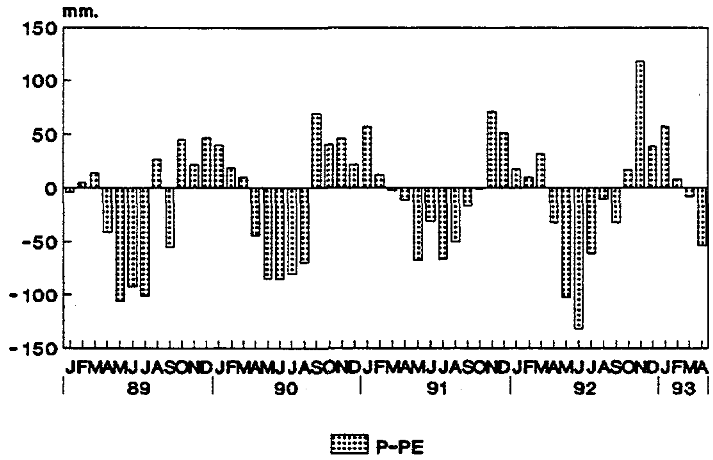

Fig. 6. Precipitation reduced by potential evaporation, Spillepeng.

sludge (about 23 mm month^{-1}) than is released through the drainage system. The water input, precipitation plus water added with the waste water treatment sludge, is compared with the landfill discharge in Fig. 7. The discharge did not respond to the large precipitation in November 1992. The leachate volume in 1992 was 257 mm, which corresponded rather closely to the water added with the sludge, 291 mm.

A detailed analysis of the discharge from the old landfill shows that the monthly peak flow, 12 mm in January 1990, occurred after an autumn which was drier than normal. The local precipitation for October 1989–January 1990 was 200 mm and the drainage volume 35 mm. Thus, the peak cannot be attributed to the rainfall in the preceding few months. The drainage volume in October 1990 was 10 mm as compared with only 6 mm in September. In this case, the peak was caused by large rainfall amounts in September (130 mm). Also, from November to December 1992 the

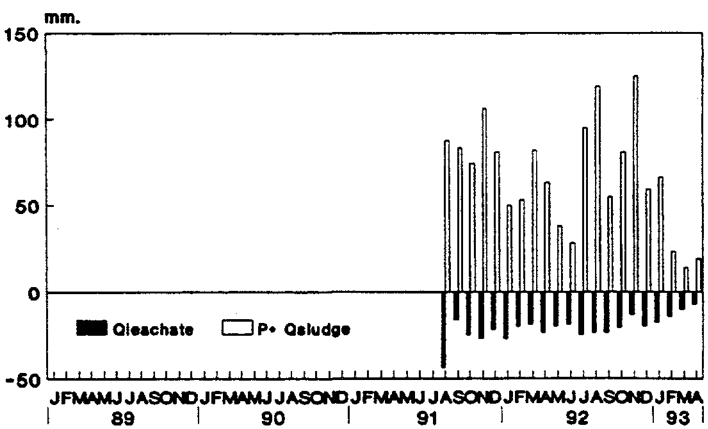

Fig. 7. Monthly water input as precipitation and waste water treatment sludge compared with discharge, SYSAV active landfill, Spillepeng.

monthly leachate production increased from 5 mm to 10 mm as a consequence of a large rainfall amount (120 mm) in November. Although there is some response to high monthly rainfall on the leachate production, there is no clear correlation.

The discharge from the old landfill was low (5–6 mm month^{-1}) in July–September 1990, October 1991 and August–November 1992. It seems that evaporation losses do not manifest themselves in reduced drainage until late summer or early autumn. The slow and very reduced response of the discharge from the old landfill to the surface processes, precipitation and evaporation, indicates that the percolation through the landfill occurs mainly as matrix flow and that macropore flow is not significant.

The monthly discharge fluctuations for the two test cells are pronounced, but the monthly drainage volumes are still small. The response to rainfall is not clear. Daily rainfalls of 30 mm are not recognized as increased drainage, not even in wet periods in the autumn when evapotranspiration does not occur. Small prolonged discharge peaks are often observed the month after high monthly precipitation, as for Test Cell 1 in September 1989, and for both test cells in November 1989, and all through the winter 1992–93. However, the very high rainfall (130 mm) in September 1990 did not produce any leachate.

The difference between the discharge hydrographs from the test cells, which show monthly fluctuations and cessation of flow, and from the old landfill, from which the flow is almost constant, indicates that some macropore flow occurs in the test cells. The main difference is, however, that the drainage volumes from the test cells are much smaller than from the old landfill. Water is continuously accumulating in the test cells.

Thus, a comparison of the drainage characteristics of the three different landfills shows that it is only in the test cells that the leachate production is, and only sometimes, influenced by high monthly precipitation. The test cells have an average height of only 5.5 m, and some macropores may still form a continuous system. In the much higher old landfill, evaporation losses in the summer do not manifest themselves in reduced drainage until in the early autumn. Increased monthly discharge is due to accumulated precipitation over many months.

7. Comparison between landfills and natural land

In a long-term perspective, a landfill is a natural part of the environment. By comparing the water balance terms for the landfills and for the two agricultural basins Höje and Värpinge, some information about how fast the landfills are approaching natural conditions, from a quantitative physical hydrological point of view, can be obtained. By taking the difference between precipitation (adding waste water treatment sludge contribution) and runoff or leachate production, the sum of evaporation losses and accumulated water storage over the year is determined. The results for several years and the various landfills are given in Table 1. Comparison is made with the Höje river discharge and the small tile-drained experimental basin Värpinge. No measurements were performed in Värpinge in the period 1989–1992; instead mean values for the period 1971–1980 were used. The annual precipitation

Table 1
Annual precipitation (p) and the sum of evaporation and accumulation of water in landfills and basins (mm)

Year	p	Old	Test Cell 1	Test Cell 2	Active	Höje	Värpinge
1989	452	358	429	436	–	369	–
1990	576	480	563	537	–	427	–
1991	638	544	633	623	–	399	–
1992	556	476	505	474	590	375	–
Mean	556	465	533	518	590	392	441

was 589 mm, the evaporation 441 mm, the drainage 113 mm and the deep groundwater discharge 28 mm.

The values in Table 1 are higher for the landfills than for the agricultural basins. It was stated above that heat conduction does not contribute to increased evaporation from landfills as compared with that from grass-covered fields. It was also shown that internal processes within the landfill have little influence on the water balance. Therefore, water is accumulating in all the landfills. As a mean, found as the difference between landfill values and Höje basin values in Table 1, 70 mm year^{-1} is accumulated in the old landfill, but the accumulation is much higher in years when precipitation is high. The accumulation in the test cells corresponds to an increased volumetric moisture content of 0.10 over 4 years.

The height of the active landfill is increased by about 4 m year^{-1}. In the previous discussion, it was shown that the moisture content of municipal waste must increase by about 0.1 for leachate to appear and by 0.25 for the landfill to reach field capacity. For a layer of 4 m thickness, these two volumetric moisture contents correspond to storages of 400 mm and 1000 mm, respectively. The sum of evaporation and increased storage given for the active landfill in Table 1 is within this range. It may thus to a large extent represent annual accumulation of precipitation and waste water treatment sludge, indicating that evaporation from the active landfill is minor. Although there is no soil cover on the active landfill, high monthly water input is not reflected in high monthly drainage. The fact that the discharge is not lower in the summer than in the winter (Fig. 7) also indicates that the evaporation is small.

The leachate production for several years is summarized in Table 2. The net

Table 2
Annual discharge, q, from test cells and from the old landfill as a whole, annual discharge regarding 30% of the old landfill as being comparable with the test cells, q^*, and annual water accumulation, Δ, for the old landfill and from the two test cells at Spillepeng (given as mm year^{-1}); the values are rounded to the nearest 10 mm

Mean	$p-e$	$q_{\text{Test Cell 1}}$	$\Delta_{\text{Test Cell 1}}$	$q_{\text{Test Cell 2}}$	$\Delta_{\text{Test Cell 2}}$	q_{old}	Δ_{old}	$q_{\text{test-mean}}$	q^*_{old}	Δ^*_{old}
1989	80	20	60	20	60	100	−20	20	135	−55
1990	150	10	140	40	110	100	50	25	130	15
1991	240	10	230	20	220	90	150	15	120	120
1992	180	50	130	80	100	90	90	65	100	80
Mean	160	30	130	40	120	95	70	30	120	40

precipitation, determined as the precipitation at the site reduced by the calculated evaporation from agricultural basins, is compared with measured annual leachate volumes. The net precipitation always exceeds the drainage volume from the test cells, but in the old landfill, water stored in previous years can contribute to higher annual drainage than precipitation excess.

Annual storage in the landfills is calculated to be 125 mm in the test cells and 70 mm in the old landfill. As the leachate production from the old landfill does not vary much from year to year, the annual accumulation of water depends very much on the precipitation volumes. In years with little precipitation, as in 1989, there is a net loss of water. The volume of drainage water from the test cells varies from year to year, but is much less than from the old landfill. Water accumulates in the young waste in the test cells also in dry years.

As 15 ha, or 30%, of the old landfill is comparable with the test cells, i.e. the waste is only a few years old, the water balance computation for the old landfill can be divided into two parts, the test cell water balance being applicable to 30% of the landfill. As the mean annual leachate from the test cells was 35 mm and the leachate from the old landfill, including the 30% of rather new waste, was 95 mm, it means that the mean leachate production from the 70% of the landfill consisting of old waste is 120 mm year^{-1} and the annual accumulation of water $(p - e - q)$ is 40 mm year^{-1}. Thus, annual storage is of importance also for the part of the landfill which consists of only old waste.

8. Conclusions

The hydrology of a landfill is greatly controlled by the soil cover in the sense that the evaporation losses and the time distribution of the percolation into the waste material depend on the thickness of the cover and on the vegetation on it. If the soil cover is not thin, the evaporation corresponds to the regional evaporation. Much water can be stored within the waste deposit. In young landfills the leachate production is minor, 30–40 mm year^{-1}, and percolation through the waste deposits occurs as preferential flow. Water is still accumulating in 10-year-old deposits. From landfills of this age, the rate at which drainage water leaves the landfill is rather constant in time, with only small, much delayed, seasonal fluctuations.

9. Acknowledgements

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Paper II

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Evaporation from an active, uncovered landfill

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Abstract

The effect on evaporation of the presence of biodegradation heat in an uncovered biocell under operational conditions is elucidated. The contribution of the heat of biodegradation was investigated by measuring the temperature gradient in the top layer of a landfill. The evaporation was calculated by combining the energy budget with an expression for sensible heat flux, and taking the atmospheric stability into account by introducing the similarity theory of Monin and Obukhov. It was found that the biological heat enhanced the net energy flux and the actual evaporation by 20% and 10%, respectively.

1. Introduction

There is a great need to understand the mechanisms that govern the infiltration of rainwater into a landfill. These include evaporation, transpiration, surface run-off and infiltration rate. The production of leachate from a landfill is controlled by the surface conditions and when considering the process over a long period of time, all water that infiltrates deep into a landfill will form leachate (Stegmann and Ehrig, 1989; Bengtsson et al., 1994). In a young landfill, however, water is still accumulating, and only a small part of the infiltrating water forms leachate. The amount of infiltrating water may be determined by a simple water budget approach as the difference between precipitation and evaporation and surface run-off. However, surface run-off has been shown to be a minor component, even when soils of very low permeability are used as cover, due to the development of fissures (Ham and Bookter, 1982; Ettala, 1987; Karlqvist, 1987; Booth and Price, 1989; Nyhan et al., 1990). Consequently, evaporation and transpiration in the case of a vegetated cover,

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constitute the most important mechanisms governing water loss from the landfill surface, and the resulting reduced infiltration.

Evaporation from a landfill with a soil cover that is not thin and has well-established vegetation is comparable to the regional evaporation. Thus, the long-term evaporation can be determined by using the water budget equation and measurements of precipitation and regional run-off (Ehrig, 1989; Bengtsson, 1994).

However, the hydrological characteristics of an active, uncovered landfill differ from the regional characteristics, and very little is known about evaporation from an operational landfill (Campbell, 1983; Ehrig, 1983). An active, uncovered landfill is characterized by both a large infiltration capacity, water storage potential and, in the case of biodegradable refuse, heat production in the top layer due to aerobic decomposition.

Due to the high infiltration capacity, the precipitation is rapidly transported into the landfill, at the expense of evaporation. Campbell (1983) estimates that the evaporation from an operational landfill surface in the UK could annually be as low as 40% of the precipitation due to the high infiltration capacity. Karlqvist (1987) also reports low evaporation and high infiltration rates from an uncovered landfill in the Malmö region. Ettala (1987) reported, from an investigation of two landfills in Finland, that when refuse was used as cover material, the infiltration rate could be as high as 60 mm h^{-1}.

Several researchers have reported the existence of an aerobic top layer in uncovered landfills. Ehrig (1982) estimates that aerobic conditions are prevailing in the upper 0.5 m in an uncovered compacted landfill. According to Christensen and Kjeldsen (1989) oxygen always diffuses into the landfill refuse, but, due to the presence of oxygen-consuming bacteria, the aerobic zone is limited to 1 m in compacted waste. Hogland et al. (1993) report oxygen concentrations of the order of 15% and temperatures of the order of 80°C at a depth of 2 m in a pilot-scale landfill consisting of compacted industrial waste covered with wood chips. In this case, the coarse structure and large fractional pore space of the industrial waste facilitated the diffusion of oxygen into the waste. Kubler (1984) made some observations in connection with wood chip piles that may be assumed to be valid also in the case of refuse material. He found that the oxygen diffusion depends mainly on the fractional pore space, since oxygen diffuses mainly through the interstitial space rather than through the solid material. The occurrence of an aerobic zone in the top layer of the biocell then depends mainly on fractional pore space. Furthermore, all oxygen that diffuses into a pile of wood chips was found to be consumed, and the heat generated was lost by sensible and latent heat flux.

The biodegradation heat may enhance the evaporation if the water supply is not limited. The effectiveness of heat originating from aerobic decomposition in evaporating moisture is pointed out by, for example, Caffrey and Ham (1974), Kubler (1984) and Miller (1991). Persson and Rylander (1977) noticed enhanced evaporation from uncovered, small-scale lysimeters when the leachate was recirculated. This was attributed to the heat produced during decomposition. Using eight small-scale lysimeters, Ham and Bookter (1982) observed an increased evaporation from lysimeters without soil cover as compared with covered lysimeters. The authors attributed this to the exposed pieces of paper at the surfaces of the uncovered lysimeters. The temperature of the refuse was also measured and the results showed that it was initially significantly higher in the lysimeters without soil cover, but decreased rapidly as the refuse decomposed. Although not pointed out in Ham and Bookter's paper, the evaporation from the uncovered lysimeters might have been enhanced by the heat of decomposition.

To summarize, there are two characteristics of an uncovered landfill that affect the evaporation in opposite directions: the high infiltration capacity, which diminishes the evaporation, because of the rapid transportation of precipitation into the landfill, and the heat production in the top layer, which contributes to the energy available for evaporation. For most active landfills the high infiltration seems to be the dominant process of the two, i.e. the evaporation from an active, uncovered landfill is lower than from a comparable, covered landfill. However, if an abundant water supply at the surface is secured by, for example, recirculating the leachate, water will evaporate at the potential rate. Due to the contribution of the heat of biodegradation, the evaporation then exceeds the evaporation rate from a comparable, covered landfill.

In this paper, the contribution of the biodegradation heat to the total energy available for evaporation is investigated for a biocell under operational conditions. The actual evaporation and the enhancement of the evaporation, due to the biodegradation heat, is also computed.

2. Theory

The landfill surface may be regarded as a system which transfers energy and mass (water vapour) between the ground and the overlying air. Since the change in heat storage is small, the net energy supplied to the surface is mainly divided into sensible heat and latent heat which is transported into the atmosphere by turbulent diffusion. These energy fluxes are governed by the turbulent mass transfer coefficients for heat and water vapour and the temperature and vapour pressure gradients between the evaporating surface and the atmospheric sub-layer.

When the water supply is abundant, that is, when the evaporation surface consists of free water or saturated soil, the evaporation rate is called potential evaporation. This term, as pointed out by Granger (1989) in his systematic review of the concept of potential evaporation, is not unambiguously defined in the literature. A variety of definitions exists, e.g. Penman (1948), Van Bavel (1966) and Priestley and Taylor (1972).

If the supply of water is limited and the rate of evaporation drops below the potential rate, a corresponding quantity of energy becomes available which affects mainly the temperature, the humidity and the turbulence of the air overlying the ground, and thus the impact on the net radiation in the energy budget will be relatively small (Brutsaert, 1982). The surface temperature rises, and if it exceeds the air temperature, unstable conditions will come into being.

According to Granger (1989), the methods to calculate the evaporation developed for non-potential conditions presuppose, in most cases, that the actual evaporation can be expressed as a fraction of a certain potential evaporation. Penman (1948), for example, assumed that the actual evaporation was proportional to the potential. However, this approach introduces some ambiguity; Brutsaert (1982) pointed out that the potential evaporation is often determined on the basis of meteorological data observed under non-potential conditions. That is, if the water supply had been abundant the potential evaporation computed would not have been the same. An approach such as Penman's can provide only an index of the actual evaporation under non-potential conditions (Granger, 1989).

Because of the ambiguity involved in the approach outlined above, a different method was chosen here. By computing the sensible heat flux, using a standard heat transport equation and taking the atmospheric stability into account by introducing the framework of the Monin–Obukhov similarity theory, the actual evaporation can be calculated as the residual in the energy budget for the evaporation surface. The method has been widely used with good results in determining regional sensible and latent heat flux, using remotely sensed surface temperatures, e.g. Soer (1980), Soares et al. (1988) and Sugita and Brutsaert (1990). Katul and Parlange (1992), using a ground-based infra-red (IR) temperature transducer, reports good agreement ($R^2 = 0.86$) between calculated evaporation and lysimeter measurements on a field of bare soil for a variety of atmospheric and soil moisture conditions.

If the surface is sufficiently large, i.e. if the contribution of energy from local advection can be ignored, the energy budget equation for the evaporation surface can be written as

$$R_n + G = Q_n \tag{1}$$

$$Q_n = H + LE \tag{2}$$

where R_n is the net radiation flux (W m^{-2}), G is the heat flux out of the refuse, Q_n is the net energy available for evaporation (W m^{-2}), H is the sensible heat flux (W m^{-2}), L is the latent heat of vaporization (J kg^{-1}) and E is the evaporation flux (kg m^{-2} s^{-1}).

G can be expressed as

$$G = Q_{bio} - S \tag{3}$$

where Q_{bio} is the heat produced by aerobic biodegradation (W m^{-2}), S is the rate of change in heat storage (W m^{-2}), taken as positive when heat is accumulating in the refuse.

R_n consists of a net short-wave and a net long-wave radiation term,

$$R_n = (1 - \alpha)R_s + \epsilon(R_l - \sigma T_s^{\,4}) \tag{4}$$

where R_s is the incoming short-wave radiation flux (W m^{-2}), α is the surface albedo, ϵ is the surface emission coefficient, R_l is the incoming long-wave sky radiation flux (W m^{-2}), σ is the Stefan–Boltzmann constant ($5.67 \cdot 10^{-8}$ W m^{-2} K^{-4}) and T_s is the surface temperature (K). Note that the long-wave term is negative, i.e. the net long-wave radiation is outgoing. The long-wave term is denoted R_{nl} in this paper.

If the net radiation is measured with a radiometer during the night, when the short-wave term in the above equation is zero, the measured flux is equal to the net long-wave radiation and the emission coefficient can easily be determined if the surface temperature is known. Solving the above equation for ϵ gives

$$\epsilon = \frac{R_n}{(R_l - \sigma T_s{}^4)} \tag{5}$$

Since the temperature gradient turned out to be almost constant with time, the upward heat flow, G, was calculated by making a steady-state assumption.

$$G = \lambda \frac{\partial T}{\partial z} \tag{6}$$

S, the change in heat storage, is given by

$$S = \int_{z=0}^{z=z_{\mathrm{const}.T}} \rho c_p \frac{\partial T}{\partial t} \mathrm{d}z \tag{7}$$

where $T(z, t)$ is the temperature in the refuse at depth z and time t; $z = 0$ is at the surface and $z = z_{\mathrm{const}.T}$ is the depth at which the temperature is constant throughout the year, λ is the thermal conductivity, ρ is the density of the top layer and c_p is the specific heat of the top layer.

The sensible heat flux is given on the basis of the Monin and Obukhov (1954) similarity theory as

$$\theta_s - \theta_a = \frac{H}{k u_* \rho c_p} \left[\ln\left(\frac{z}{z_{0h}}\right) - \psi_h(\xi) \right] \tag{8}$$

$$V = \frac{u_*}{k} \left[\ln\left(\frac{z}{z_0}\right) - \psi_m(\xi) \right] \tag{9}$$

where θ_s and θ_a are the potential temperatures at the surface and in the air at height z, respectively, k is Von Karman's constant (here taken as 0.4), u_* is the friction velocity (m s^{-1}), ρ is the density (kg m^{-3}), c_p is the specific heat of the moist air (J kg^{-1} K^{-1}), and V is the windspeed. The reference level $z = 0$ is taken as the basis of the roughness elements, z_0 and z_{0h} are the roughness lengths for momentum and sensible heat, respectively. In the absence of wind velocity profile measurements, z_0 is commonly expressed as a fraction of h_0. Here $z_0 = h_0/10$. The roughness length for sensible heat can be estimated by

$$z_{0h} = 7.4 z_0 = \exp[-2.46(z_{0+}^{1/4})] \tag{10}$$

where $z_{0+} = (u_* z_0)/\nu$ is the roughness Reynolds number and ν is the kinematic viscosity (Brutsaert, 1982).

If the change in barometric pressure is assumed to be small, the difference $\theta_s - \theta_a$ can be replaced by $T_s - T_a$. ψ_h and ψ_m are the similarity stability correction functions for sensible heat and momentum, respectively, of Monin and Obukhov (1954) which depend on $\xi = (z - d_0)/L$, where L is the Obukhov length.

$$L = \frac{-u_*^3 \rho}{kg\left[\left(\frac{H}{T_a c_p}\right) + 0.61E\right]} \tag{11}$$

where g is the gravitational constant.

Several functional stability correction functions are found in the literature. Of these the Businger–Dyer formulations are commonly used (Sugita and Brutsaert, 1990; Katul and Parlange, 1992). For unstable conditions they can be written as

$$\psi_h = 2\ln\left[\frac{(1+x^2)}{2}\right]$$

$$\psi_m = \ln\left[\frac{(1+x)^2(1+x^2)}{(1+x_0)^2(1+x_0^2)}\right] - 2\arctan(x) + 2\arctan(x_0) \tag{12}$$

where $x = (1 - 16\xi)^{1/4}$ and $x_0 = (1 - 16z_0/L)^{1/4}$.

In order to compute E for each time step by solving Eqs. (1), (6) and (8), an iteration procedure has to be employed. Initial values of u_*, z_{0h}, H and E are given by assuming $\psi_h = \psi_m = 0$.

3. Description of the experimental site and the measurements

The measurements were conducted in the active landfill at the Spillepeng disposal site located in Malmö in the south of Sweden. The site is managed by SYSAV, the Southwestern Scania Solid Waste Company. The active landfill consists of three biocells with a total area of 5 ha. The biocells are built up in layers of 2 m. The total height at the time of measurement was 10 m. For sanitary reasons and to make it stable enough for sanitation trucks to drive on, each layer is covered with a thin layer of wood chips. The use of soil is more common in Sweden but SYSAV has chosen to use wood chips because the influence on gas and water flow is expected to be less. Due to compaction of the top layer with a conventional steel-wheeled compactor, the $z = 5$–20 cm layer can be characterized as a mixture of refuse and wood chips, while the upper 5 cm layer consists of wood chips. At depths exceeding 20 cm the material consists of refuse only. The geometrical height of the roughness of the wood chip elements was estimated to be 5 cm. The landfill is located on boulder clay and the leachate is collected in a drainage system. The bio-gas is also collected. The refuse is mixed municipal solid waste (MSW), compacted to a density of 0.7, and waste water treatment sludge. This sludge contains 95% water.

The temperature in the top layer was measured every 6 h during the period 16th July to 30th November 1993, using thermocouple connected to a data logger. The temperature was measured at nine positions randomly distributed over an area of 250 m^2, at depths of 0, 5 and 15 cm. The temperature measured at $z = 0$, which is the base of the roughness elements, was taken as the surface temperature. This simplification is justified by the fact that the elements are not densely placed. At one location, the temperature was also measured at a depth of 45 cm, making a total of 28 measuring points.

Using data from hourly radiometer measurements of the incoming and outgoing radiation and the ground surface temperature collected over 5 days, the surface emissivity and the albedo, as a daily average, were calculated to be 0.6 and 0.3, respectively. The radiation was measured with two Siemen Ersking radiometers. The lower half of one of the radiometers was equipped with a black-body radiator of known temperature.

A total of 33 samples, with an approximate weight of 5 kg each, were taken from the top layer (0–20 cm) on four occasions during the measurement period. A constant high moisture content, 30 vol%, was observed. In the laboratory, a large number of samples were mixed and from this mixture a 10 kg sample was taken and compressed to its original density, 0.63. Using this sample, the field capacity was determined to be 29%. Thus, it seems that the water content in the top layer of the biocell was constantly at field capacity.

Leachate data were provided by SYSAV and precipitation was measured at Sjölunda sewage plant, Malmö. The data on air temperature, vapour pressure of the air and radiation data (incoming short- and long-wave) were provided by the Swedish Meteorological and Hydrological Institute, SMHI.

4. Energy flux

An average temperature for each depth was calculated on a daily basis. These are plotted with the daily average air temperature in Fig. 1.

The figure clearly shows a temperature gradient and thus indicates the existence of

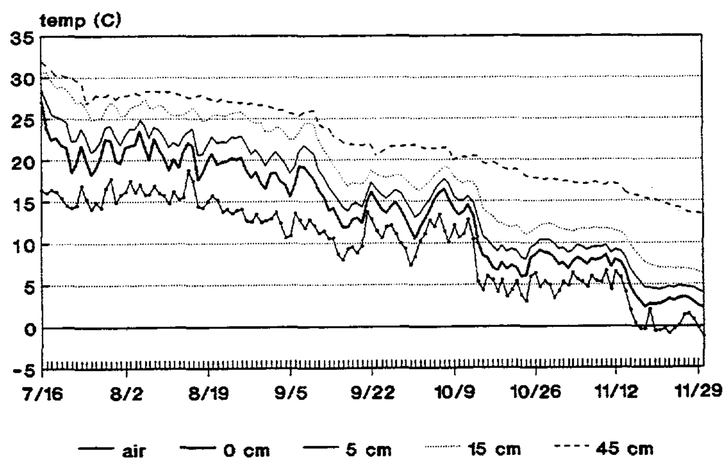

Fig. 1. Temperatures of the air and at the depths of 0, 5, 15 and 45 cm.

an upward heat flow. Since heat is being generated at all levels of the aerobic top layer, the temperature gradient is not linear but is steeper towards the surface. The gradient at the surface is in the range 0.5–1 K cm^{-1}, with the highest gradient occurring during the summer months and the lowest in November.

According to the temperature measurements made regularly by the staff at Spillepeng, the temperature at a depth of 2 m is constant at 35°C (K. Engblom, personal communication, 1993). Accordingly, the change in heat storage within the waste was calculated for the upper 2 m. Temperatures between the depths of 45 cm and 2 m were interpolated linearly. Savage (1989) has reported the specific heat for refuse-derived fuel pellets (RDF), containing no water at all, to be on average 1.6 kJ kg^{-1} K^{-1} and the thermal conductivity to be 0.16 W m^{-1} K^{-1}. Since RDF pellets are made of municipal solid waste which has been sorted, shredded and compacted, these figures may represent an average property, under dry conditions, of the materials of which MSW is composed. The specific heat of the different layers was calculated by using the appropriate proportional figures for the specific heat for the constituents wood, refuse and water. The volumetric heat capacity, ρc_p, for wood chips at a density of 0.63 and refuse at a density of 0.7, both with a volumetric water content of about 30%, was found to be of the same order, 2 MJ m^{-3} K^{-1}. This is, of course, a rough estimate of the volumetric heat capacity, but, as will be shown later in this report, the change in heat storage is small compared with the other terms in the energy balance.

To calculate the upward heat flux in the ground, the thermal conductivity, λ, has to be determined. According to thermal experiments on swine waste, consisting of faeces, uneaten feed, bones, plastics, paper, glass etc., conducted by Mears et al. (1975), the thermal conductivity of a compost material varies linearly with its water content (in weight%). Extrapolation of the curve to 100% water content gave a result close to the thermal conductivity of water. Extrapolating the curve to 0% gave a thermal conductivity of 0.24 W m^{-1} K^{-1}. Mears concluded that the thermal conductivity for different water contents could be determined by a single measurement at a known water content. By using this point and the value for 100% water the entire relationship could be established.

The dry solid fraction of the refuse was assumed to have a thermal conductivity of the order of 0.20 W m^{-1} K^{-1}, which is the average of the values given above for RDF and swine waste. By using this value and the thermal conductivity of water, 0.65 W m^{-1} K^{-1}, the thermal conductivity of the refuse in the top layer with a water content of 48% by weight (30% by volume) was determined, using the method of Mears as outlined above, to be 0.40 W m^{-1} K^{-1}.

The surface temperature was found to be higher than the air temperature not only on a daily average basis but also when the midday values were compared, that is, due to the biodegradation heat, unstable conditions prevailed during the entire measurement period. Taking this into account, the sensible heat was calculated on a daily basis using the Monin and Obukhov similarity theory, Eqs. (8), (9) and (11). The latent heat was determined as the residual in Eq. (2).

The other components of the heat budget were also calculated on a daily basis. The result is shown as weekly averages in Fig. 2.

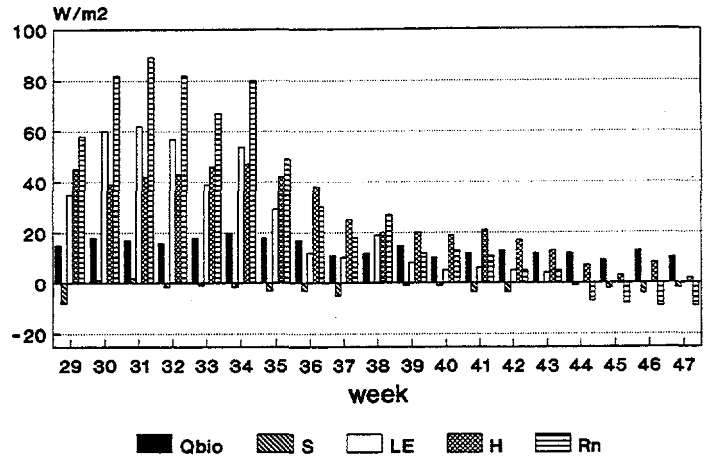

Fig. 2. The heat budget components as weekly averages.

As can be seen in the figure, the net short-wave radiation dominated the energy balance during July and August (Weeks 29–34). About 20% of the net energy supplied to the evaporation surface during this period was made up of biodegradation heat. Since the net long-wave radiation (outgoing) and the biodegradation heat were almost constant with time, the relative significance of these terms became greater in the autumn when the net short-wave radiation decreased. Considering the whole measurement period, the change in heat storage was minor and negative, i.e. there was a net loss of heat in the refuse. In November the net radiation became small and negative and the energy budget was dominated by the biodegradation heat. The heat available for evaporation, Q_n, was found to be equally divided into sensible and latent heat until the beginning of September (Weeks 35–36) when the portion of Q_n becoming sensible heat increased. In November (Weeks 44–47) the total amount of energy supplied to the surface was lost as long-wave radiation and sensible heat.

Since R_{nl}, Q_{bio}, S, H and LE are mutually related, it is difficult to estimate the influence on the energy budget of the biodegradation heat. If Q_{bio} were non-existent, this would affect primarily the surface temperature, which would decrease, that is, R_{nl} would increase (the outgoing long-wave radiation would decrease), and H and LE would decrease. S could be assumed to be negligible.

This condition was roughly simulated by excluding the biodegradation term from the energy budget and assuming the new surface temperature to be equal to the air temperature on a daily average basis. A simulated net energy flux for conditions with no contribution from biodegradation heat, Q_n^*, was calculated and compared to the existing Q_n. Fig. 3 shows Q_n and Q_n^* calculated on a daily basis and plotted as weekly averages.

It can be seen that the contribution to the energy budget from the biological heat was clearly significant and, as already mentioned, constant over time. Its relative importance grew during the measurement period. During the first 6 weeks of the measurement period, the increase in the energy budget due to the biological heat

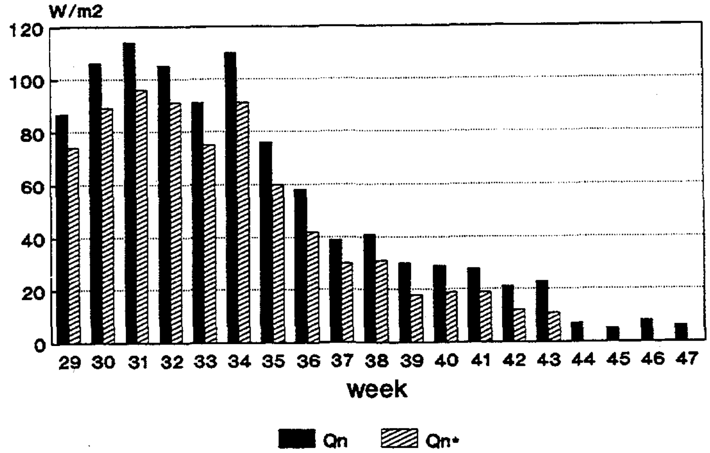

Fig. 3. Q_n and Q_n^* plotted as weekly averages.

was in the range of 10%, after which it steadily increased up to 50%. During the last 4 weeks, the net energy flux would have been zero without the contribution from the biological heat. Regarding the entire measurement period, the biological heat enhanced the net energy flux by about 20%.

Because of the complicated relationship between R_{nl}, Q_{bio}, S, H and LE it is not

Table 1
Results: the terms of the energy budget are given as weekly averages (W m^{-2}) and the evaporation as weekly totals (mm)

Week	R_{ns}	R_{nl}	Q_{bio}	S	H	LE	E	α
29	89	−31	15	−8	45	35	9	1.06
30	115	−33	18	1	39	60	16	1.10
31	124	−35	17	2	42	61	16	1.07
32	118	−36	16	−2	43	57	15	1.07
33	102	−35	18	−1	46	40	11	1.09
34	126	−46	20	−2	47	55	15	1.09
35	86	−37	18	−3	42	28	8	1.13
36	63	−33	17	−3	38	12	3	1.14
37	46	−28	11	−5	26	11	3	1.11
38	51	−24	12	0	20	19	5	1.18
39	43	−31	15	−1	20	8	2	1.23
40	33	−20	10	−1	19	5	1	1.18
41	46	−35	12	−3	21	6	2	1.17
42	54	−49	13	−4	17	5	1	1.28
43	30	−25	12	0	13	4	1	1.46
44	18	−25	12	−1	7	0	0	–
45	8	−16	9	−2	3	0	0	–
46	17	−26	13	−4	8	0	0	–
47	11	−20	10	−2	2	0	0	–
Sum							108	

possible to calculate directly the proportion of the actual evaporation that can be attributed to the biological heat. However, by using the Penman equation as an index of the actual evaporation, a rough estimate of the amount by which the actual evaporation is enhanced by the presence of the biological heat can be made by making the following assumption

$$\alpha = \frac{E}{E^*} = \frac{E_p(Q_n, T_a, e_a, u)}{E_p(Q_n^*, T_a, e_a, u)}$$

where α is the enhancement factor, E is the actual evaporation, E^* is the evaporation if there were no contribution from biodegradation heat and E_p is the potential evaporation as defined by Penman. α was calculated on a daily basis.

The energy budget and the results of the evaporation calculations are summarized in Table 1.

Note, since the computed actual evaporation was zero during Weeks 44–47, calculation of α was not considered to be meaningful.

5. Water budget

The result of the evaporation calculation was evaluated by establishing a water budget for the measurement period. In Fig. 4, the water input, precipitation + water content in the sludge, is plotted cumulatively together with the amount of leachate and the calculated evaporation. The water storage was calculated as the residual in the water budget.

The production of leachate is constant at about 5 mm week^{-1} and, except for Week 40 when the leachate production rose to 8 mm, no response to rainfall could be seen. Considering the whole measurement period, precipitation and water content in the sludge accounts for 340 mm and 30 mm, respectively, leachate for 90 mm and

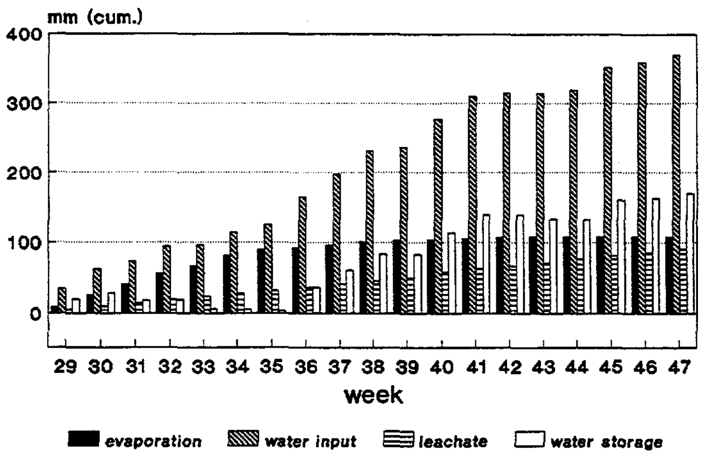

Fig. 4. Evaporation, water input, leachate and water storage plotted cumulatively on a weekly basis.

evaporation for 110 mm. Thus, the net water input during the measurement period was 260 mm. This leaves 170 mm for storage in the landfill. As a comparison, Bengtsson et al. (1994) found that the average annual accumulation of water in 4 year-old test cells was 125 mm when the average annual net water input, i.e. P–E, was 160 mm and that the rate of accumulation was higher during years with high precipitation. These figures correspond to an annual accumulation of about 45 mm m^{-1} in the biocell and 20 mm m^{-1} in the test cells (an average height of 6 m was assumed).

6. Conclusions

Although the net radiation dominates the energy budget, the heat originating from the aerobic degradation in the top layer of a landfill makes a significant contribution. The upward heat flux due to biological heat is always present, maintaining a potential for evaporation throughout the year. Since the heat flux is almost constant over time, its relative importance increases as the net radiation decreases. Assuming that the measurement period is representative of a whole year, it can be concluded that the net energy flux and the actual evaporation is enhanced by 20% and 10%, respectively.

Acknowledgements

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Paper III

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Accumulation of water and generation of leachate in a young landfill

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Abstract

Seven-year-old pilot-scale landfills were investigated with respect to operation of the soil cover, accumulation of water and rainfall–leachate flux relationship. The annual soil moisture fluctuation in the soil cover was of the order of 120 mm. Given an average precipitation of 580 mm, the net water input into the landfill, split into storage and leachate discharge, was on average 250 mm annually. The portion that becomes leachate increased from zero to about half during 6.5 years. The storage increase was about 180 mm/m during this period. Fair agreement between the calculated storage and the moisture content in solid waste samples was observed. A 1–2 month time lag in the net water input–leachate discharge relation was observed. It is suggested that the observed short leachate flux response to a period of infiltration may indicate that a large part of the water is retained by surface tension as thin films in restricted channels, which is overcome by gravity when the input is suddenly increased. The storage–leachate flux relationship was quantified and was found to consist of a family of curves depending on the storage and the rise or decline of leachate flux. © 1997 Elsevier Science B.V.

Keywords: Water; Accumulation; Leachate; Landfill

1. Introduction

With the objective of investigating the environmental impact of landfills, a great body of scientific work has been published during the last two decades, proposing different models for predicting the field-scale arrival of polluted leachate through the lower boundaries of the landfill. Today, when gas and leachate are collected, the focus of the hydraulic research task also includes investigating the presence and mobility of water in the landfill interior. Not only is water necessary for the first step in the biodegradation process, the hydrolysis, a high moisture content also facilitates the redistribution of nutrients and microorganisms within the landfill (Christensen and Kjeldsen, 1989; Augenstein and Pacey, 1991). The moisture content is probably the most important factor governing the biochemical processes in a landfill (e.g. Augenstein and Pacey, 1991; Ehrig, 1991), but also the water flux has been shown to enhance the biodegradation process (Klink and Ham, 1982).

One of the common approaches in modeling the generation of leachate is to regard a landfill as a spatially lumped system and calculate the produced leachate volume using the continuity equation (Fenn, 1975; Kmet, 1982; Campbell, 1983). The water

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budget method, one of the earliest models, was proposed by Fenn (1975) and was further developed by Kmet (1982). By using a combination of 25 different methods for calculating the different terms in the water budget and comparing the results with the measured leachate discharge at five different landfills, Lu et al. (1981) and Gee (1983) showed that the method is uncertain and can be marred by very large errors. The existing water budget models assume that the waste is at the field capacity and that a certain water input produces an equal amount of outflow at the lower boundary.

The most widespread and well-known model is the computer model HELP reported by Schroeder (1983). This model takes into account the initial accumulation of water up to the field capacity and the time lag in the precipitation–leachate discharge relation by calculating the flow rate through the landfill. However, as pointed out by Farquar (1989), since the HELP model assumes a uniform flow field, neither the water budget method nor the HELP model can handle the phenomenon of fast preferential flow or channeling. Preferential flow is most significant in young deposits due to the coarser structure, and has been observed in several investigations (Harris, 1979; Walsh and Kinman, 1979; Blakey, 1982; Ham and Bookter, 1982; Holmes, 1983; Korfiatis et al., 1984; Blight et al., 1992; Bengtsson et al., 1994). Stegmann and Ehrig (1989) refer to investigations where dug-up landfills have shown evidence of preferential flow. As the refuse biodegrades and settles, the medium gets more homogeneous, the dry density increases and the voidage volume decreases, which, in turn, limits the fast preferential flow to some extent in older deposits. The extent of preferential flow is not only dependent on the structure and voidage volume but also on the precipitation rate. Additional flow routes are developed during periods of high infiltration rates (Jasper et al., 1985). The existence of preferential flow is very important and is believed to be the reason why existing models are not in agreement with actual field observations (Ehrig, 1983; Stegmann and Ehrig, 1989).

Knowledge of the space and time variability of the water storage in landfills is lacking to a great extent. The spatial variability of water content is large, ranging from saturated to dry conditions. Dry spots and regions in waste deposits have been reported by several researchers, e.g. Harris (1979). The long-term storage of water in a landfill is either held by surface tension or impermeable layers creating hanging water tables. The maximum volume of water per unit volume of landfill material that can be held by capillary forces against gravity is defined as the field capacity. This concept originated in soil science and is somewhat vague, since the gravity drainage may take a long time to occur. Further, there is no standard method to determine the field capacity. Since the water volume held as hanging water tables is probably small compared with the field capacity, the latter will be used to denote the maximum water content that on a long-term basis can be stored in a landfill. According to a summary by Bengtsson et al. (1994) of the data given in the literature, the initial volumetric moisture content is in the range of 0.15–0.20 (corresponds to a moisture content of 35 wt.% (dry) and a wet density of 0.5–0.8), the field capacity is of the order of 0.40 and the increase of moisture content before leachate occurs is of the order of 0.10. The time it takes for the moisture content to increase from its initial content to field capacity can be significant. The rate of accumulation depends on waste composition and age, initial moisture content, density, presence of macropores and preferential flow (Blakey, 1982; Holmes, 1983). Bengtsson et al. (1994) found that water was still accumulating in 10-year-old deposits. Holmes (1983) performed extensive experiments on samples of domestic refuse collected in situ in order to investigate the absorptive capacity. He categorized the samples in three groups according to their age, 3.3, 9.6 and 15.5 years, and found that all samples had a potential for further moisture uptake. He noted further that the moisture content for the oldest group was lower than the middle aged group and attributed this phenomenon to moisture release as the basic structure of the absorbing medium degraded. The dependence of storage capacity on changes in the structure of the organic material due to degradation was pointed out by Ehrig (1983). Holmes (1983) found that the field capacity decreased with age, which is in agreement with the observations of Blight et al. (1992).

It is reasonable to expect that a positive correlation between storage and leachate production exists, which also has been explicitly pointed out by Fungaroli and Steiner (1971) and Blakey (1992).

To summarize, the dynamics of water accumulation

Table 1
Waste composition and density for pilot-scale landfills #1, #2, #4 and #5 (Nilsson et al., 1992)

Landfill	Composition	Wet density
1	mixed household- and industrial-waste	0.46
2	mixed household-, industrial-waste and sewage sludge (5%)	0.50
4	household waste	0.65
5	mixed household waste and sewage sludge (5%)	0.57

have been reported sparsely in the literature and needs to be investigated for several reasons. The time before a landfill reaches field capacity is significant and must be accounted for when the leachate production is predicted. Thus, the water budget equation should be combined with a storage–leachate production relationship in order to solve the continuity equation for leachate production. Further, the moisture content for a landfill, even when given as a lumped value, constitutes an important input to models that describe the biochemical processes in terms of leachate quality and gas production.

This paper is a hydrologic investigation of four pilot-scale landfills during their accumulation phase. The operation of the soil cover, the accumulation dynamics, and the storage and leachate relationship are investigated.

2. Description of field site and measurements

The pilot-scale landfills investigated in this study were built by SYSAV, the waste management company in the very south-west of Sweden. The landfills are located at Spillepeng disposal site in Malmö and cover an area of 1600 m^2 each. The average depth is 6 m: 10 m at one end, sloping at 15% down to 2 m at the other end. A plastic liner underlies the waste. The leachate is collected and the volume is measured weekly. The pilot-scale landfills were originally built to investigate the biogas production and leachate quality as a function of waste composition. There are six pilot-scale landfills, but only the data from four of them, designated as #1, #2, #4 and #5, will be used in this study. This is because the leachate collection system in #3 is suspected to be malfunctioning and that leachate is recycled in #6.

According to Nilsson et al. (1992) the disposal of waste started in November 1988 in landfills #1 and #2 and in May 1989 in landfills #4 and #5. Landfills #1 and #2 were covered with a 0.5 m thick clay cover during April–May 1989, and landfills #4 and #5 during September 1989. During August and September 1990 the landfills were covered with additional 0.3 m plant soil. However, recent measurements in situ showed that the soil cover is about 1 m thick.

Grass was sown in October 1990, and the vegetation had become established by the summer of 1991 (Nilsson et al., 1992).

The waste composition of the landfills is given in Table 1.

Solid samples have been taken annually by drilling. Two holes were drilled in each cell and samples were taken at two depths: 2 m and 4 m measured from the lower boundary of the soil cover. In each bore hole two samples were taken from each depth, which makes a total of eight samples from each cell. The set of four samples that were taken at the same depth were mixed into a single sample for which the moisture content was determined.

Four lysimeters were installed beneath the soil cover in October 1993 to measure the water that percolates through the soil cover. Four neutron probe pipes were installed close to the lysimeters to measure the moisture content in the soil cover. Surface runoff was measured during October 1993 to March 1994. However, runoff was registered only at two occasions and was less than 10 mm. It was concluded that surface runoff had a minor influence on the water input to the landfills; therefore, no further measurements were made. Surface runoff has been observed to be negligible even when soils with low permeability are used as cover, due to the development of fissures (Ham and Bookter, 1982; Ettala, 1987; Karlqvist, 1987; Booth and Price, 1989; Nyhan et al., 1990).

3. Operation of soil cover

The net water input into the waste domain of a landfill is governed by the operation of the soil cover. Given a certain rainfall event the operation of

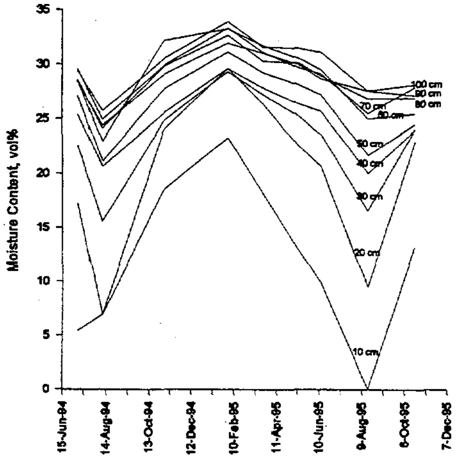

Fig. 1. The Soil moisture content in the soil cover at the depths 10–100 cm.

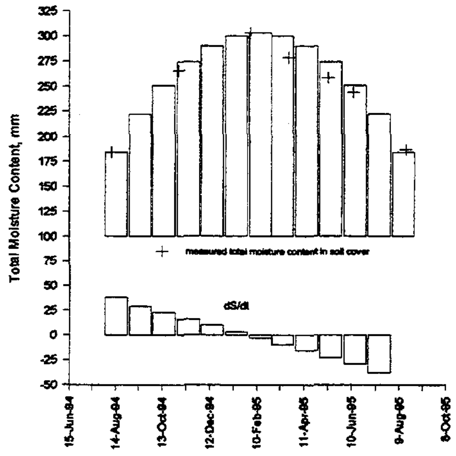

Fig. 2. The total moisture content for the soil cover S_s, and the monthly change in storage dS_s/dt (mm).

the soil cover includes smoothing of the time distribution of the rainfall, due to hydraulic resistance and storage capacity of the soil cover, and reduction of the total rainfall volume due to evapotranspiration which extracts water from the soil.

The neutron probe data were converted to soil moisture by using a calibration curve that had been established for the actual soil. In Fig. 1 the average data is plotted for the time period 7th July 1994 to 23rd October 1995.

A clear seasonal dependence can be seen, with the lowest value after the summer in August and the highest in January and February. The shallow levels show large fluctuations, whereas the soil moisture content is consistently high at deeper levels, showing only minor fluctuations. This is in agreement with observations made by Booth and Price (1989) who reported from an investigation of the operation of a landfill cover in Illinois that only the soil moisture in the upper 0.3 m showed a seasonal dependence. The soil moisture data for different depths was lumped together to a total value and expressed as a water height in millimeters. A symmetrical discrete curve was fitted to the data and the result is shown in Fig. 2.

The total moisture budget on an annual basis is 119 mm, and the changes in moisture content are almost linear with respect to time.

Evaporation from a covered landfill is dependent mainly on the type and extent of the vegetation on the soil cover. For a landfill with a soil cover that is not thin and with well-established vegetation, evaporation is comparable with regional evaporation (Ehrig, 1989). The annual evapotranspiration from the pilot-scale landfills was assumed to be comparable with the evaporation landfills from the adjacent small agricultural river basin Höje (Bengtsson et al., 1994). Hence, assuming that the deeper groundwater accumulation is negligible, the annual actual evapotranspiration for the landfills was determined as the difference between precipitation and Höje River discharge. The actual evaporation for the landfills was estimated on a monthly basis by using the potential evaporation according to Penman's definition as an index indicating the fraction of the annual actual evapotranspiration that could be attributed to a certain month. The potential evaporation according to Penman's definition was given by the Swedish Meteorological Institute, SMHI, as average monthly value calculated for a grass vegetated surface in Malmö for the years 1931–1974.

4. Flux and accumulation in the waste domain

The net water input I to the waste domain was calculated on a monthly basis using a simple water budget:

$$I = P - ET - \frac{dS_s}{dt}$$

where P is the precipitation, ET is the evapotranspiration, and dS_s/dt is the change in storage in the soil cover. The figures on dS_s/dt which are given in Fig. 2 were applied to the entire measurement period. The estimation of ET and the total soil moisture budget is fairly accurate on an annual basis. The figures on ET and dS_s/dt on a monthly basis represent only a mean behavior and the condition during 1 year of the measurement period respectively. When calculating I the errors in ET and dS_s/dt may offset each other or cooperate in either direction, which may give anomalies as a negative I. Since the mean behavior of I was considered to be fairly accurate, the effect of the errors of ET and dS_s/dt was compensated by smoothing of I by a moving average with a 3 month window. The change in storage in the waste domain dS_w/dt was calculated as

$$\frac{dS_w}{dt} = I - Q_l$$

where Q_l is the average leachate production from the four landfills. The value of S_w is the storage volume in millimeters exclusive of the initial unknown moisture content. The result is shown in Fig. 3.

The first 1.5 year period will be excluded from further investigation. However, it can be pointed out that leachate production started immediately as the waste was disposed. A response to the water input during the winter 1989/1990 can also be observed. When the soil cover was completed in August–September 1990 and vegetation was established on the spring 1991 the leachate production was almost zero until October 1991 when the first leachate started to appear. The accumulated storage was then about 60 mm m^{-1}. The leachate production showed a clear seasonal dependence with an exponential increase in peak values with respect to time. However, even in 1995, still only half of the net water input formed leachate, which can be seen in Table 2. The base flow was negligible, although showing a small

Table 2

Annual precipitation P, infiltration I, storage in the waste domain dS_w, and leachate production Q_l

	P	I	dS_w	Q_l (mm)
1989 April–	391	59	46	13
1990	576	201	185	16
1991	640	245	238	7
1992	558	236	183	53
1993	630	264	191	73
1994	664	284	163	121
1995 –Nov	432	254	139	115
Average	580	250		

increase during 1994 and 1995 it did not contribute more than about 3 mm/month.

The accumulation of water dS_w/dt typically took place from October to April, whereas there was a 3–5 month weak recession period during the summer. This is also in agreement with the lysimeter observations, where no infiltrating water could be detected during June to September.

The accumulation of water in the waste was highly correlated to the net water input until October 1994, when dS_w/dt suddenly decreased. This may be attributed to the landfill approaching field capacity, and a larger portion of the net water input becomes leachate. According to the figures given earlier in this paper, the increase of the moisture content from initial moisture content up to field capacity is roughly assumed to be 0.25, which, given an average height of the landfills of 6 m, equals 1500 mm of total storage.

4.1. Results of solid sampling

The calculated figures on accumulated water in the landfills is here compared, on an annual basis, with data on moisture content determined by solid sampling. The mean and standard deviation for the solid sample data are plotted together with the calculated total moisture content in Fig. 4. Annually, there is a total of eight data points from the four test cells. Since the samples were distributed, the moisture content is expressed in wet weight percent. The total moisture content was determined by adding the standard value of initial moisture content that was given earlier, 0.35 wt.% (dry) or 0.26 wt.% (wet), to the previous calculated accumulation of water S_w. The mass loss

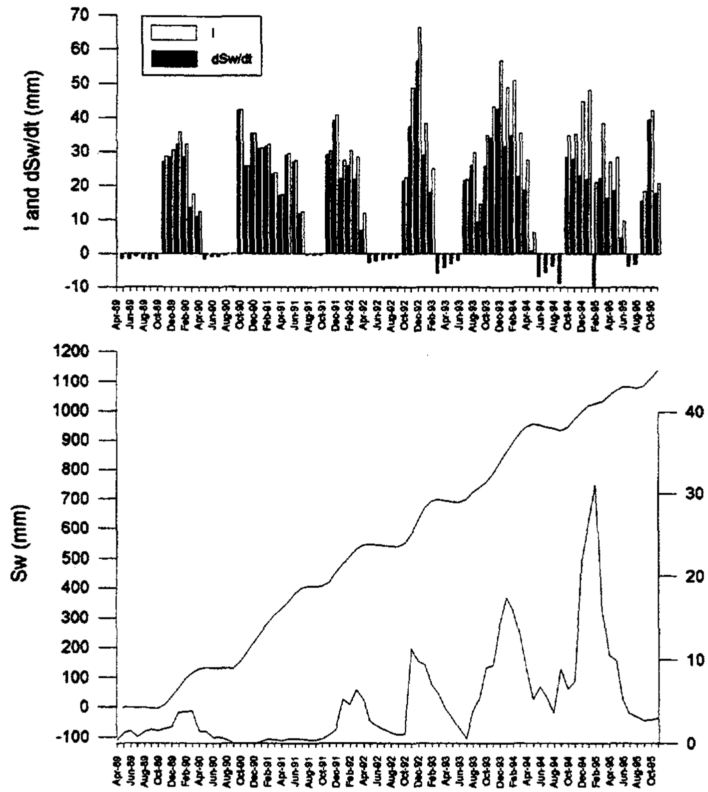

Fig. 3. I, dS_w/dt, S_w and Q_l are plotted along the time axis.

due to biogas production, which was about 7% of the total initial mass of the disposed waste, taken over the entire measurement period, had to be taken into account when calculating the moisture content in wet weight percent.

As expected, the solid sample data show a large variance due to the heterogeneity of the landfill formation. The large standard deviation indicates that a greater number of data points is required in order to determine accurately the mean moisture content by sampling. However, the agreement between the sampled data and the calculated moisture content is fair.

4.2. Storage–discharge relationship

The leachate peaks occurred during the winter months of January to March, except for 1992 when the peak was already in November. It is seen in Fig. 3 that the peak of the response experiences a time lag of about 1–2 months compared with the peak of the net water input. The response by the waste domain to the water input is a short leachate pulse with the peak in the middle of the actual accumulation period. After the peak, the leachate flux declines rapidly, even though the water input continues. This behavior may first seem like an anomaly, but it can probably be

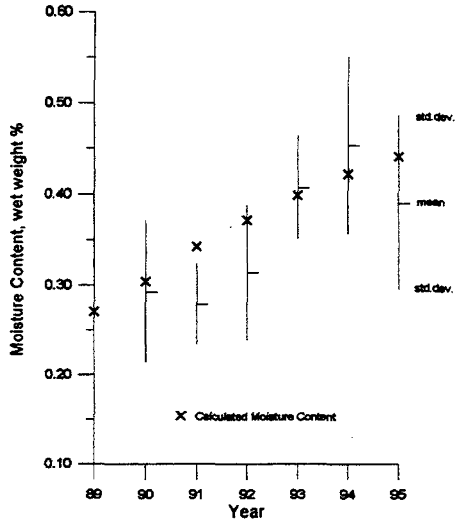

Fig. 4. The calculated total moisture content and the mean and standard deviation for the solid sample data.

explained by the geometrical properties of the refuse. Owing to the composition and the coarse structure of municipal solid waste, it is likely that a dominant part of the water movement takes place as a moving film on surfaces. Ferguson (1993) proposed a hydraulic model where all water movement and retention take place as mobile respective static films held by surface tension. Given this conceptualization, the observed response of the waste domain can be explained as follows. As water is introduced to the waste domain, it moves slowly down as a thin water film on solid surfaces. The surface tension creates a static water film which spreads and increases in thickness up to the point where the tension forces barely exceed gravity. When the waste system is triggered at this point with an input peak, which makes the thickness of the static water film increase rapidly, the gravity overcomes the surface tension and a large part of the retained water rapidly propagates down along preferential pathways.

It is difficult to establish a storage–leachate flux relationship for a landfill during its accumulation phase for two reasons: (1) each accumulation period will have a unique storage–leachate flux relationship; (2) considering a certain accumulation period, the leachate flux–storage relation will be double valued. Thus, the long-term storage leachate relationship is made up of a family of storage discharge relationships, depending on the storage and the rise or decline of the leachate flux. This is shown in Fig. 5, where the storage and leachate flux data for the years 1991–1995 are plotted. The total storage has been divided by the average depth of the landfills and is expressed in millimeters per meter. The data points were separated in rising and recession periods and a second degree polynomial, $S_w = a + \beta_1 Q + \beta_2 Q^2$, was fitted, showing high coefficients of determination.

A power function, $S_w = aQ^b$, was fitted to the peak values and to the average storage and leachate flux for each pair of rising and recession curves. The average curve is plotted in Fig. 6 together with data given by Blakey (1992).

The landfills approach the field capacity and a change in the storage–flux pattern is expected. The appearance of the last couple of rising and recession curves may be due to the approaching of the point where the waste domain reaches steady-state. At steady-state the curve will form a narrow loop which starts and ends at the same point. Blakey (1992) used four test cells of 3.5 m height and 87.6 m^2 area to investigate infiltration and accumulation of water in domestic waste. The cells were monitored over 5 years and about 280 mm of water infiltrated into the waste annually, which, as shown in Table 2, is comparable with the net water input in this study. In Fig. 6, data on infiltrating water and leachate production from Blakey's reference cell, containing household waste, have been used to establish a storage–discharge relationship. Owing to the large averaging interval, 6–12 months, of the given data effects such as lag or response fluctuations are lumped together into a single value. The last value deviated significantly from an otherwise clear trend. Although not pointed out by the author, the deviation is likely to be due to clogging in the drainage system. Its effect was offset by the use of the average of the two last values.

The relationships show significant consistency. Blakey's test cell shows, however, a little more response to an increment in storage, which may be attributed to the smaller depth of the waste formation and a slightly higher net water input.

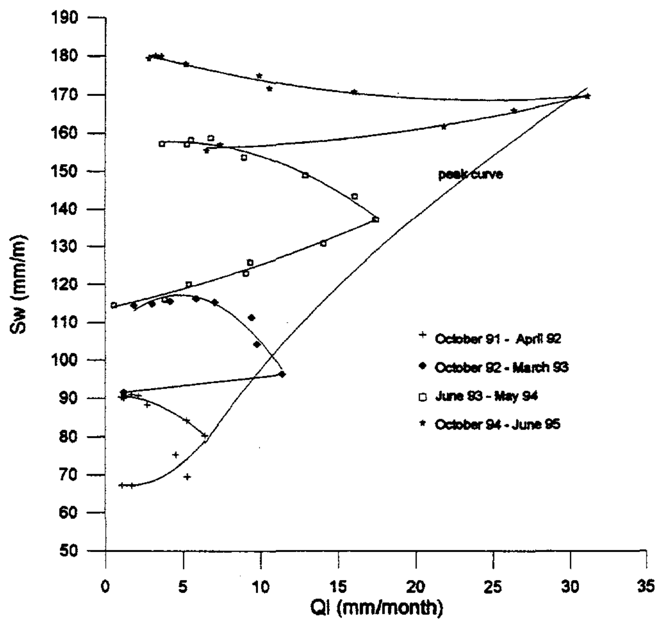

Fig. 5. Storage S_w (mm m^{-1}) plotted as a function of leachate flux Q_l (mm/month) together with power function fitted to the peak values.

Table 3
Fitting result $S_w = a + \beta_1 Q_l + \beta_2 Q_l^2$

	Period							
	11/91–3/92	3/92–10/92	10/92–11/92	11/92–6/93	7/93–1/94	1/94–8/94	10/94–2/95	2/95–11/95
a	68	90	—	107	114	156	156	183
β_1	−1.50	0.39	—	4.41	0.93	0.95	−0.11	−1.19
β_2	0.49	−0.31	—	−0.48	0.024	−0.11	0.018	0.024
R^2	0.74	0.98	—	0.93	0.98	0.98	0.99	0.96

Table 4
Fitting result $S_w = aQ_l^b$

	Peak value	Average value	Blakey (1992)
a	3.43	47.0	13.9
b	0.50	0.51	0.84
R^2	0.97	0.99	0.99

The results from the curve fitting procedures are summarized in Tables 3 and 4.

With the exception of the first period, it is shown that the storage and the leachate flux is highly correlated and that the relation can be described by simple polynomial and power functions, explaining over 90% of the variance.

Regarding that the landfill is a prototype system,

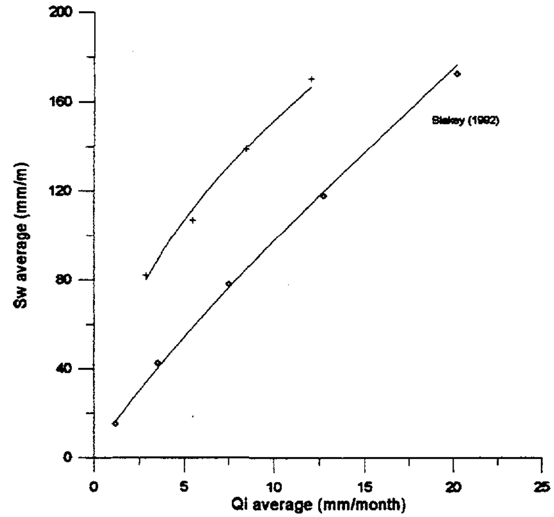

Fig. 6. Comparison with data given by Blakey (1992).

far-reaching conclusions cannot be drawn from this curve fitting procedure. However, the similarities in the results presented here and the data given by Blakey (1992) suggest that the average behavior in the storage–flux relation may be universally applicable.

5. Summary and conclusions

The storage of water shows a seasonal dependence. Accumulation of water takes place typically during October to April. A weak recession occurs during May to September, when almost no water percolates through the soil cover. An average of 250 mm of water infiltrates annually. The water input is split into storage and leachate discharge. During the 6.5 year period since the landfills were built, the portion of the water input to become leachate has increased from zero to about half and the moisture content of the waste has increased 180 mm m^{-1} from its initial value. During the last period of infiltration a clear indication of the landfill approaching field capacity can be seen: dS_w/dt decreased steadily in favor of increased leachate production.

The leachate flux is characterized by a negligible base flow and a sharp peak in the middle of each accumulation period as response to the water input. The flux declines rapidly before the infiltration period ends. This behavior is attributed to the retention regime in the refuse, where water is held by surface tension and may abruptly be overcome by gravity.

The storage–leachate production relationship is constituted by a family of relations which may be described by simple polynomial functions. The average relation for storage and flux or storage and flux peak values can be described by power functions. Good agreement in comparison with the data given by Blakey (1992) suggest that there may exist a universally applicable storage–leachate flux relationship for a waste medium.

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Paper IV

Hydrological Characteristics of Landfills - Implications for Modeling

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Abstract The hydrologic characteristics of landfills, such as evaporation, function of the soil cover, accumulation dynamics, the hydraulic properties of the waste medium and the existence of fast flow in restricted channels, are reviewed. Existing lumped parameter and continuum models are presented. Based on the observed landfill characteristics, a physical conceptualization of the internal geometry of a landfill is presented, together with a kinematic wave model for channel flow.

INTRODUCTION

Leachate and biogas emissions are the main environmental problems associated with landfills containing biodegradable matter. The leachate that escapes the landfill is subject to a variety of retention mechanisms in the under laying strata and its effects are only local [Christensen, 1992], whereas the emission of methane gas has a impact on the global scale since it is a green house gas. In order to stabilize a landfill in a controlled and efficient way, so that the environmental impacts are minimized, understanding of the processes in the landfill interior is crucial. A landfill containing biodegradable waste can be regarded as a biological reactor where water plays an important role. The first step in the anaerobic degradation process, in which the organic elements are hydrolyzed and fermented to fatty acids, alcohols, carbon dioxide and hydrogen, is important and may be rate limiting for the whole degradation process [Leuschner and Melden, 1983]. Since water is the only carrier of substances within a landfill, a high water content facilitates the redistribution of nutrients and microorganisms within the landfill [e.g. Augenstein and Pacey, 1991; Christensen and Kjeldsen, 1989]. Furthermore, water flux has been shown to enhance the biodegradation process [Klink and Ham, 1982]. Åkesson and Nilsson [1997] investigated the seasonal variability of leachate production and quality from two pilot-scale landfills, each containing 4000 tonnes of waste. They found that a number of leachate quality parameters were correlated to the leachate flux such that the biochemical conditions in the landfill interior appeared to alternate between methanogenic, at low flux rates, and acidogenic, at high flux rates. The authors attributed this phenomenon to spatially variable biochemical conditions.

Several researchers have pointed out that, in order to improve the existing models for biodegradation and leachate quality, further research must focus on the presence and flux of water [Augenstein and Pacey, 1991; Ehrig, 1983; Straub and Lynch, 1982]. The flow field in landfills is highly nonuniform. Water is flowing in structural voids that form a continuous system of channels that "short cuts"a large bulk of the landfill. The importance of this phenomenon is undisputed [Ehrig, 1983; Stegmann and Ehrig, 1989; Zeiss and Major, 1993] but it has not been fully investigated. Zeiss and Major [1993] points out the need to revise or extend the Darcian flow models to take the channeling phenomenon into account.

The objective of this study was to compile the hydrological characteristics of municipal solid waste (MSW) landfills by reviewing the literature, and to evaluate their implications in the modeling of water movement and leachate generation.

This paper is organized as follows. First the hydrologic characteristics of landfills are described with reference to field and laboratory observations. The main focus here is on the characteristics of the landfill waste domain. The function of soil covers including infiltration, storage, and evaporation is qualitatively outlined. The existing models are reviewed and discussed in terms of underlying assumptions and how the landfill characteristics have been taken into account. Based on field observations, a conceptualization of the internal geometry is proposed. Finally, given the characteristics of landfilled waste, the implications for modeling are discussed and a kinematic wave model for channel flow is presented.

Matters not considered in this study are the transport of chemicals and microorganisms, including growth and decay mechanisms.

HYDROLOGICAL CHARACTERISTICS

The precipitation-leachate flux relation

The input of water to a landfill system is determined by precipitation and the surface conditions. The rainfall-leachate discharge relationship for a young landfill is dominated by the accumulation of water and the degree of channel flow in the landfill. The amount of leachate produced is small and may show large fluctuations due to fast channel flow [Bengtsson et al., 1994]. The leachate production from an old landfill, in which the storage capacity has been fully utilized, shows seasonal variation and is comparable to regional runoff in that respect [Bengtsson et al., 1994; Ehrig, 1989; Karlqvist, 1987]. Surface runoff has been found to be small, due to fissures in the soil cover, even when low-permeability soils are used [Bendz et. al., 1997; Booth and Price, 1989; Ettala, 1987; Ham and Bookter,1982; Karlqvist, 1987; Nyhan et al., 1990]. The abstractions of water during the precipitation-leachate production process is thereby limited to evaporation, storage in the soil cover and storage in the waste domain.

Evaporation

Evaporation from a covered landfill is mainly dependent on the type and extent of the vegetation on the soil cover. For a landfill with a soil cover that is not thin and has well established vegetation, the evaporation is comparable to regional evaporation [Bengtsson, 1994; Ehrig, 1989], thus, it does not constitute a special landfill characteristic and will not be discussed here. However, the evaporation from an active, uncovered landfill is different due to the existence of an aerobic top layer [Caffrey and Ham, 1974; Christensen and Kjeldsen, 1989], the high infiltration capacity [Campbell, 1983; Karlqvist, 1987; Ettala, 1987], and the lack of transpiration. The latter two phenomena facilitate a rapid transport of water into the landfill and thus reduce the amount of evaporable water in the surface layer, while the former increases evaporation due to significantly greater contribution to the energy budget. Taking atmospheric stability into account, Bendz and Bengtsson [1996] found that the heat originating from the aerobic degradation in the top layer of an active, uncovered landfill enhanced the net energy flux by 20% and the actual evaporation by 10%.

Function of soil cover

The way in which a soil cover functions is dependent on the soil characteristics and the thickness of the cover. A large amount of literature is available on the functioning of soil covers [e.g. Booth and Price, 1989; Karlqvist, 1987; Nyhan et al., 1990] therefore, only a small number of pertinent features of the soil cover will be outlined here. Following the input of a certain volume of water, the soil cover will cause smoothening and a time lag due to storage effects and the hydraulic resistance of the soil, and a certain loss due to evapotranspiration which extracts water from the soil. In a hydrological investigation of four pilot-scale landfills in the south of Sweden, with an annual precipitation of about 600 mm, Bendz et al. [1997] found that the soil moisture content in the 1-m-thick soil cover with well-established vegetation was seasonally dependent. The annual soil moisture budget was 120 mm. Most of the soil moisture fluctuations took place in the upper part of the cover, while the soil moisture content at the bottom was constantly high, showing only minor fluctuations. This is in agreement with the results reported by Booth and Price [1989] from an investigation of the function of a landfill cover in Illinois. They found that only the soil moisture in the upper 0.3 m showed a seasonal dependence.

Hydraulic properties of waste medium

The values of the saturated hydraulic conductivity of landfilled waste reported in the literature show large variations due to spatial variability, different measurement scales, degree of compaction, particle size and measurement methods. The range of the values given stretches from 10^{-2} to 10^{-8} ms^{-1}, according to a review by Bleiker et al. [1995]. The higher values represent waste with a low degree of compaction and measurements performed *in situ* by pumping tests since they contain a large horizontal component. The structure of fresh refuse is coarse, but as biodegradation proceeds the refuse homogenizes and settles, the void volume decreases, and the density increases [Holmes, 1983]. Because the oldest refuse is located at the deepest levels in the landfill and each layer is compressed by subsequent waste emplacement, the compaction increases with depth. This was observed by Oweis et al. [1990], Bleiker et al. [1995] and Burrows et al. [1997] together with an accompanying decrease in hydraulic conductivity.

There is a serious lack of knowledge concerning the space and time variation of the water storage in landfills. The spatial variation in water content is large, ranging from saturated to dry conditions. Dry spots and dry regions in waste deposits have been reported by several researchers, e.g. Harris [1979]. The long-term storage of water in a landfill is either due to capillary forces, surface tension or impermeable layers creating hanging water tables. Burrows et al. [1997] pointed out the similarity between a modern landfill and a young sedimentary basin, and applied conventional hydrogeologhical methods to investigate the hydraulic properties of landfilled waste. The whole range of aquifers; perched, unconfined, artesian, and leaky, was found either physically or through pumping tests.

The capillary potential in refuse is likely to be small in fresh household waste due to the coarse structure. Stegmann and Ehrig [1989] stated that there was no capillarity in fresh compacted waste, but as the waste decomposes the capillarity of MSW may reach 350 mm. Korfiates et al. [1984] established a moisture retention curve for 6-month-old MSW by fitting a curve to experimental data. The curve yielded a capillary potential of 350 mm at the time of disposal when the initial water content of the waste was assumed to be 0.175 volumetric percent. However, individual refuse elements, such as a bundle of paper or a piece of wood, may show a higher capillarity and a large storage potential. It is reasonable to assume that the water absorbed in these refuse elements will not participate in the flow process on a short time basis. The maximum volume of water per unit volume of landfilled waste that can be held by capillary forces against gravity is defined as the field capacity. This concept originated in soil science and is somewhat vague since gravity drainage may take a long time to arise. Further, there is no standard method of determining the field capacity. The HELP model defines the field capacity as the volumetric water content at a capillary pressure of 0.33 bar and uses a default value of 0.292 [Schroeder et al., 1994]. According to a summary by Bengtsson et al. [1994] of the data given in the literature, the initial volumetric waste moisture content is in the range of 0.15-0.20, the field capacity is of the order of 0.40 and the increase in waste moisture content before leachate is produced is of the order of 0.10. The time taken for the moisture content to increase from its initial value to field capacity, denoted t_{FC}, can be significant. The rate of accumulation depends on composition and age of the waste, the initial moisture content, density, presence of voids, and channel flow due to high precipitation intensities [Blakey, 1982; Holmes, 1983]. Bengtsson et al. [1994] found that water was still accumulating in 10-year-old deposits.

Holmes [1983] performed extensive experiments on samples of domestic refuse collected in situ in order to investigate the absorptive capacity. He categorized the samples in three groups according to their age, 3.3, 9.6 and 15.5 years, and found that all samples had a potential for further moisture uptake. He noted further that the moisture content in the oldest group was lower than that in the intermediate group and attributed this to release of moisture as the basic structure of the absorbing medium became degraded. The dependence of storage capacity on changes in the structure of organic material due to degradation was pointed out by Ehrig [1983]. Holmes [1983] found that the field capacity decreased with age, which is in agreement with the observations of Blight et al. [1992]. Using small-scale lysimeters 1.2 m in depth, Gee [1983] and Ham and Bookter [1982] reported that the cells were at field capacity after 9 months and 500 mm precipitation. Rosqvist et al. [1997] presented a hydrological study of a 4-meter-deep pilot-scale landfill containing fresh household waste to which irrigation of 30 mm was applied daily. The cell was covered with a tarpaulin so no evaporation took place. Leachate production started already after five days. After less than 20 days, i.e. 600 mm of water input, steady-state conditions were reached corresponding to an increase in water storage of about 10 volumetric %.

Storage-leachate flux relation

It is reasonable to expect a positive correlation between storage and leachate production [Fungaroli and Steiner, 1971; Blakey, 1992]. Bendz et al. [1997] determined the storage-leachate flux relation for four pilot-scale landfills during the first 6.5 years after the landfills had been completed. It was found that the storage-leachate flux relation involved a time lag of about 1-2 months and that the relation consisted of a family of relations depending on the storage and whether the leachate flux was rising or falling.

Channel flow

Theoretically, no leachate can be produced until the landfill has reached its field capacity. However, due to the heterogeneity and anisotropy of the waste medium, the field capacity is spatially variable, and some parts of the landfill therefore reach field capacity long before the landfill as a whole does. Water may be flowing in locally saturated regions although most of the landfill may be well below field capacity. Furthermore, the internal geometry of a landfill facilitates the downward flow of water in restricted channels and voids. Stegmann and Ehrig [1989] referred to investigations where excavated landfills have shown evidence of channel flow. Channel flow, which is most significant in young deposits due to their coarser structure, has been observed in several investigations [Bengtsson et al., 1994; Blakey, 1982; Blight et al., 1992; Burrows et al., 1997; Harris, 1979; Ham and Bookter, 1982; Holmes, 1983; Korfiates et al., 1984; Walsh and Kinman, 1979]. Zeiss and co-workers [Zeiss and Major, 1993; Zeiss and Uguccioni, 1995; Uguccioni and Zeiss, 1997] were the first to quantify the proportion of a compacted waste column that was made up of channels. This was done on a large laboratory scale using columns filled with raw MSW equipped with grids of sensors installed at three different levels. The flow channels were found to constitute about 25% of the cross-sectional area. They also found that the channel flow was very fast. Following a wetting event, water was registered at the outflow within 15 minutes. Rosqvist and Bendz [1998] conducted tracer experiments in a large undisturbed sample from a 22-year-old pilot-scale MSW landfill. It was found that the expected residence time of a solute with steady irrigation of 35 mmh^{-1} corresponded to a water volume displacement of 6.1%, expressed as a fraction of the total volume of the sample. This was taken as a rough estimate of the solute transport volume [Jury et al., 1991]. When irrigating in pulses, averaging 35 mmh^{-1}, the solute transport volume increased to 13.9%. It was concluded that solute transport volume was not only a function of the internal geometry, but also of the water input pattern. This is in agreement with a study by Jasper et al. [1985] who suggested that additional flow routes may be developed during periods of high infiltration rate. As the refuse biodegrades and settles, the dry density increases and the void volume decreases which, in turn, limits the fast channel flow to some extent in older deposits.

EXISTING MODELS

Lumped models

In 1975 the U.S. Environmental Protection Agency (EPA) presented a tool to predict the volume of leachate produced. This was the water balance model (WBM) by Fenn et al. [1975]. The landfill was treated as a spatially lumped parameter system and the leachate production was calculated using the continuity equation. The model was further developed by Kmet [1982]. The most famous and widespread model for the prediction of leachate production is the computer model HELP first reported by Schroeder et al. [1984] and later in version 3 in Schroeder et al. [1994]. The HELP model, in contrast to the WBM, takes into account the accumulation of water up to field capacity and the time lag in the precipitation-leachate discharge relation. The water input to the landfill is calculated by solving the water balance equation for the surface. The water balance parameters considered are: rainfall, snow melt, evaporation, surface- and snow storage, and surface runoff. Evaporation is calculated with the modified Penman method of Ritchie [1972], and the surface runoff with the USDA Soil Conservation Service curve-number method together with a routine that takes the slope and the slope length into consideration. The landfill is divided into several subsystems and the water movement through the landfill is calculated by a routing procedure using Darcy's law combined with an expression for unsaturated hydraulic conductivity of the Brooks and Corey [1966] type. The model does not explicit take channel flow into account. However, in the latest version [Schroeder et al., 1994], inspired by the work of Zeiss and Major [1993], the option of choosing a waste layer with lower porosity, lower field capacity and lower wilting point has been added to the model in order to implicitly account for channeling.

Continuum models

Rasmuson [1978] determined the characteristic curve for screened composted household waste, measured the saturated hydraulic conductivity, and applied the unsaturated flow theory to water movement. He determined the unsaturated hydraulic conductivity using the methods of Marshall [1958], Millington and Quirk [1961], Green and Corey [1971] and Mualem [1976], and found good agreement between the different computational schemes. The initial water content and water input pattern were shown to be of great importance for the movement of water. Straub and Lynch [1982] presented a simple well-mixed reactor model and a continuum flow model for the transport of contaminants and applied it to experimental data reported in other investigations. The same mathematical framework for a continuum model was employed by Demetracopoulus [1986] but another numerical scheme to solve the differential transport equations was proposed. Lee et al. [1991] modeled water flow and leachate quality by conducting leaching experiments on shredded municipal solid waste of four different ages. Vincent et al. [1991] used an artificial model waste in the laboratory and performed flow and solute experiments to model the flow and transport processes. Ahmed et al. [1992] presented a two-dimensional numerical model for water flow through landfills during unsteady conditions, including the calculation of evaporation, runoff and infiltration. The governing Richards equation was solved for moisture content at various grid points by employing a finite difference scheme. Simulation showed that a leachate mound develops at the bottom of the landfill creating a lateral flow towards the collection system.

Straub and Lynch [1982a] and Korfiatis et al. [1984] found that infiltrating water moved downwards as wetting fronts. Gravity was considered to be dominating, whereas capillary forces were of minor importance and could be neglected when the moisture content exceeded the field capacity. This was attributed to the large pore structure of the waste, which made large suction heads impossible.

From a hydraulic perspective, the dominating approach is to regard the waste medium as a soil with special characteristics, as explicitly proposed by Canziani and Cossu [1989]. The studies referred to above have all been made at laboratory scale and treat the waste as a homogeneous, porous medium. Richards equation and the convection-dispersion equation, CDE, has been employed to describe the movement of water and the transport process, respectively. Klute's definition of soil water diffusivity and the forms of hydraulic conductivity and capillary potential suggested by Clapp and Hornberger [1978] have been employed. In none of the studies the criteria for the analysis of transport with a convective-dispersive process was investigated. Only Vincent et al. [1991] performed solute experiments, but the solute concentration was only measured at the effluent. This alone does not give any information on which process assumptions might be appropriate, since any transport model, no matter what the process assumptions might be, can be calibrated to fit the measured concentration curve if it is only measured at one depth [Jury and Roth, 1990].

Straub and Lynch [1982] and Korfiates [1984] took into account the heterogeneity and channeling effects on an ad hoc basis by simply adjusting the dispersivity parameter (molecular diffusion was regarded as negligible) and the exponent in the Clapp Hornberger [1978] expression for tension head, respectively. Quantitative descriptions of the waste in terms of composition and geometry are, to a great extent, lacking in the models referred to above, which must be regarded as a serious drawback due to the prototype character of the landfill system.

Uguccioni and Zeiss [1997] evaluated two different approaches to take the channel phenomenon into consideration: first on an ad hoc basis, by decreasing the field capacity parameter and increasing the saturated hydraulic conductivity parameter in the HELP model and, second, by using a more physically sound representation of the waste media by introducing a dual-domain flow model for a fractured porous medium (PREFLOW) calibrated to MSW. The PREFLOW model routes the flow through the matrix using the Richards equation, and if the water input rate exceeds the saturated hydraulic conductivity of the matrix, the excess water flows down the channels according to Poiseuille's law. Thus, laminar flow and circular channel cross sections are assumed. The lateral exchange of flow between a channel and the matrix is assumed to obey Darcy's law. It was concluded that although the HELP model can be calibrated to predict the early breakthrough times, by decreasing the field capacity parameter, there is a contradiction in doing this and simultaneously predict the redistribution and storage of moisture in the matrix. The PREFLOW model is physically superior, but has a major drawback in that the specification of parameters such as channel diameter, length, etc., is difficult.

All models referred to thus far have been developed at the laboratory scale. A framework for scaling up the models such that they remain valid on the landfill scale, where the spatial variability must be taken into account, has not yet been proposed.

INTERNAL GEOMETRY AND FLOW PATTERN

The geometry of the landfill interior and channel flow are important and should be taken into account when developing models for flow and transport processes. Based on the characteristics of a landfill given previously, the heterogeneity and the flow regime are qualitatively characterized here. In the next section the implications for modeling are outlined.

A landfill can be divided into three subsystems: the soil cover, the unsaturated waste domain, and, in most cases, a saturated waste domain at the bottom. The saturated waste domain arise due to direct contact with groundwater, or as the result of a bottom liner and a leachate drainage system with inefficient capacity so that the water level rises in the landfill. As pointed out earlier, hanging water tables or perched aquifers embedded in the unsaturated domain are also present. The unsaturated waste domain constitutes the largest part of the landfill and is usually of the order of 10 m thick. Depending on the landfill philosophy, the cover may be designed either to prohibit or permit a certain amount of infiltration. A cover of the latter type is made up of soil and its thickness is usually approximately 1m. The type of soil to be used commonly depends on the availability in the region. In order to restore the landfill area for recreational use and to reduce infiltration into the waste, vegetation is established on the soil cover.

As with natural materials, the internal geometry can be largely explained if the formation process is known. Thus, before discussing the internal geometry of a landfill, the construction procedure will be briefly described. The usual procedure is to construct a landfill in layers of approximately 2-3 m, either in stages, by the disposal of several thin layers, each layer being compacted, or by depositing the whole layer at one time and compacting it as a unit. The latter method gives a lower degree of compaction. The landfill is usually covered at the end of each working day with a thin layer, 0.1-0.3 m, commonly of inert masses. This is done for sanitary reasons and to make the working surface suitable for vehicles. However, the daily covering, or intermediate layers, can act as water and gas barriers inside the landfill [Lagerqvist, 1986; Blight et al., 1992]. For this reason, wood chips are sometimes used as daily covering material, as this material is believed not to influence gas and water movements in the landfill.

Although heterogeneity and anisotropy in landfills have been observed by several researchers [e.g. Oweis et al., 1990], a quantitative description of their spatial variability is lacking. The waste domain is heterogeneous and anisotropic in both horizontal and vertical directions. Due to the disposal and compaction procedure, strong horizontal stratification can be observed. Therefore, in the horizontal plane the waste domain can be assumed to be isotropic, since any possible anisotropy is negligible in comparison with the anisotropy in the vertical direction. Thus, the spatial variability of a landfill can be fully described in two dimensions, one horizontal length scale (x) and one vertical (z).

The landfilled waste medium can be conceptualized physically as discretely hierarchical, being composed of porous lens-shaped elements with partially impermeable surfaces (compacted and partially torn refuse bags). The lenses are separated by a network of voids or channels. The domains, in which flow and transport take place, can be easily distinguished as the interior of the

lenses and the structural voids. Accordingly, the heterogeneity of a landfill can be categorized into the following three length scales:
(1) The lens scale, which is of the order 0.5 m in the horizontal plane and 0.1 m in the vertical plane.
(2) The truck-load scale, characterizing the length scale of a group of lenses *in situ*, originating from the same refuse truck load (they can be assumed to share certain characteristics), which is of the order of 10 m in the horizontal plane and 1 m in the vertical plane.
(3) The landfill scale, which is of the order 100 m in the horizontal plane and 10 m in the vertical plane.
The geometry and the heterogeneity scales of two layers, separated by a daily cover, are illustrated in Fig. 1.

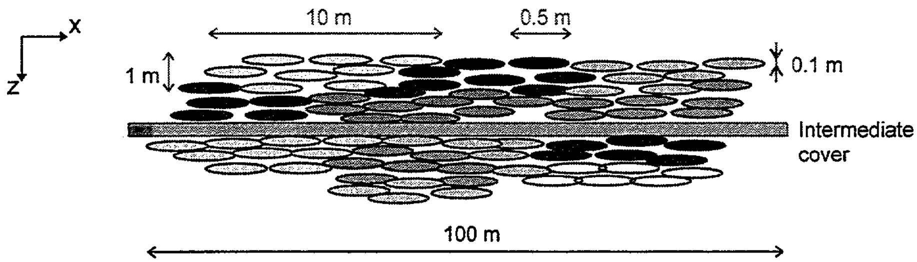

Fig.1 Geometrical configuration of two layers, separated by a daily cover in an MSW landfill. The figure shows refuse lenses (compacted and partially torn refuse bags), originating from different garbage truck loads, separated by structural voids.

Apart from this spatial variability, the internal geometry of a landfill also has a temporal variability, due to the degradation process. The structure of fresh refuse is coarse, but as biodegradation proceeds the refuse homogenizes and settles, the void volume decreases and the density increases [Holmes, 1983].

Regarding the internal geometry of the landfill, the channel flow pattern can be described as follows: Due to stratification, a significant portion of the flow takes place in the horizontal direction. These flow paths at different levels are connected by vertical short cuts. This leads to a network of flow paths, similar to those in fractured rocks or fissured media. A schematic illustration of this flow pattern in two dimensions is given in Fig. 2. In addition to this, there is also slow diffusional water movement in the matrix.

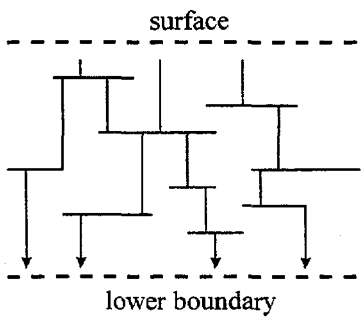

Fig.2. Schematic illustration of the flow paths in the landfill interior

A network flow model is defined by the hydraulic properties of the network members. The geometric properties, conductance, connectivity, and flow path-matrix flow transition are key parameters. Modeling of flow in fractures has focused on the geometry of the fracture network, connectivity fracture [Küpper et al.. 1995], and the hydraulic properties of the fractures such as fracture wall roughness and fracture aperture [Nordqvist et al., 1992; Moreno et al., 1988; Tsang and Tsang, 1987], and fracture skin [Thoma et al., 1992]. The spatial variability of the landfill can be considered by treating the properties of the channels as stochastic functions see, for example, Cacas et al. [1990] and Moreno and Neretnieks [1993].

The flow in the channels can be assumed to be laminar and move as a thin film on solid surfaces. Gravity dominates and viscosity is the only opposing force. The capillary force is considered negligible. Such flow is often called creeping flow, and is characterized as a moving fluid where the inertia is dynamically insignificant compared with viscosity [Sherman, 1990]. Slender viscous films with free surfaces constitute a special case of creeping flow.

IMPLICATIONS FOR MODELING

The landfill may be discretized into two domains, allowing separate assumptions to be made for flow and transport processes in each domain. As shown by Gerke and Genuchten [1993], who summarized the literature on dual-domain models for heterogeneous porous media, the dual-domain concept has been widely used when modeling flow and transport processes in soil and rocks exhibiting various kinds of heterogeneities, such as macropores, fractures and fissures. In all of these cases, significant deviations from a uniform flow field, due to fast flow in restricted pathways, has been observed.

In the matrix domain, the capillary force is considered significant and the water movement is diffusive. Given that the scale is sufficiently large, for a representative elementary volume to exist, the flow in the matrix domain can be described by Richards equation. Modeling of unsaturated flow in porous media employing Richards equation is well documented in the literature and will not be discussed here. Instead, the aim is to recognize the overlooked phenomenon of channel flow and provide a framework for modeling. The channel domain is defined by the flow path network shown in Fig.2. Rosqvist and Bendz [1998] and Zeiss and Major [1993] have verified experimentally that the channel domain constitutes only a minor fraction of the total volume. The water in the channel domain is in constant contact with the much larger water volume in the matrix domain and the two domains may be coupled by exchange terms that describe the interaction of the domains with respect to moisture and solutes.

Flux law

Due to apparent similarities it is suggested here that the framework developed by Germann and co-workers for macropore flow in soils may also be appropriate for modeling of water flow in the channel domain in landfilled waste. Germann [1990] questioned the adequacy of the potential flow theory in describing flow in porous media, since it presupposes a perfect correlation between hydrodynamic dispersion and capillary potential. Three types of cases, derived from laboratory or field observations, constitute deviations from this correlation. The fast flow in macropores constitutes one of these deviations since the change in capillary potential propagates slower than water. Beven and Germann [1981] and Chen and Wagenet [1992] proposed a power function as the macroscopic flux law. The following expression was suggested by Beven and Germann [1981] as the governing flux law for water flow in macropores:

$$q = bw^a \tag{1}$$

where q is the water flux density (ms^{-1}) in the macropores per unit cross-sectional area of soil, w is the volumetric water content, a is a dimensionless exponent and b is the macropore conductance (ms^{-1}) which can be interpreted as the lumped effect of the surface, geometrical, and spatial characteristics of the flow path. The amount of water in the macropores is expressed as a volume fraction of the soil, w, rather than the thickness of a water film. For macropore flow the parameter a can be expected to be in the range $2 \leq a < \approx 8$ [Germann and Di Pietro, 1996). where $a=2$ indicates laminar flow in a cylindrical pipe (Poiseuille's law), and $a=3$ indicates flow in planar cracks. Higher values of a indicate increasingly tortuous flow and completely dispersive flow corresponds to $a > \approx 10$. Turbulent flow is indicated when $a<2$. In Chezy and Manning equations for overland flow the exponent is equal to $^3/_2$ and $^5/_3$, respectively. Germann and Di Pietro [1996] concluded that the exponent a reflects both the properties of the static system, such as the internal geometry of the medium, and the flow process itself.

CHANNEL FLOW MODEL

The flow path network illustrated in Figure 2 is composed of a large number of channel elements. A framework is outlined below for the analysis of unsteady channel flow in one dimension with kinematic wave theory. The proposed model is applicable to a single member of a channel network or to the case where spatial variability is neglected and the network is lumped together into a one-dimensional channel domain. Water may enter a channel through the end , either at the interface between the waste domain and the soil cover or in the network connection points, or laterally from the matrix. In a flow path network model the outflow hydrograph from one channels constitute the inflow hydrograph to another.

Several researchers have applied kinematic wave theory to the vertical movement of soil moisture [Sisson, 1980; Singh and Joseph, 1993; Yamada and Kobayashi, 1988], but Smith [1983] was probably the first to develop a complete kinematic wave model. The application of the theory was extended to include infiltration and drainage into and from soil macropores [Germann, 1985; Germann and Beven, 1985]. The kinematic wave model is simple and allows the powerful method of characteristics by which a partial differential equation can be reduced into a system of ordinary differential equations. As shown by Singh [1997], the method provides analytical solutions for many cases. The model has been shown to perform well in describing the time-space history of infiltration fronts and general soil moisture patterns [Germann, 1985; Germann and Beven, 1985; Germann et al., 1996; Smith, 1983; Yamada and

Kobayashi, 1988]. These investigations have in common that they have dealt with the case when water is entering the system through the upper boundary. Here, the kinematic wave model is derived for two cases: (I) when the water enters the channel only from the upper end and (II) when the water enters laterally through the boundaries of the channel. The mass balance in the channels can be written:

$$\frac{\partial w}{\partial t}+\frac{\partial q}{\partial z}=\begin{cases} r(z,t), & 0\leq t\leq T, \quad z_A\leq z\leq z_B \\ 0, & T<t, \quad 0<z \end{cases} \tag{2}$$

where r is the lateral exchange (s^{-1}) between the matrix and the channel domain, $[z_A, z_B]$ is the interval where the exchange occurs, t is the time (s), T is the duration of the lateral inflow, and z is the depth (m). Substitution of the flux expression (1), which assumes that q is a function of w alone, into equation (2) yields the following kinematic wave equation:

$$\frac{\partial w}{\partial t}+c\frac{\partial w}{\partial z}=\begin{cases} r(z,t), & 0\leq t\leq T, \quad z_A\leq z\leq z_B \\ 0, & T<t, \quad 0<z \end{cases} \tag{3}$$

in which

$$c=\frac{\partial q}{\partial w}=abw^{a-1} \tag{4}$$

where c is the wave celerity (ms^{-1}). Equation (3) can be reduced to the following system of characteristic equations:

$$\frac{dw}{dt}=\begin{cases} r(z,t), & 0\leq t\leq T, \quad z_A\leq z\leq z_B \\ 0, & T<t, \quad 0<z \end{cases} \tag{5}$$

$$\frac{dz}{dt}=abw^{a-1} \tag{6}$$

Equation (6) describes the time and space history of the characteristic carrying the water content defined by (5) and the initial and boundary conditions.

For a square pulse input, q_u, starting at $t=0$ and ending at $t=T$, and entering through the upper end of the channel, the following initial and boundary conditions can be assumed:

$$w(z,0)=f(z), \qquad 0\leq z\leq Z \tag{7}$$

and

$$w(0,t) = \begin{cases} 0, & t \le 0,\, t \ge T \\ w_u, & 0 < t < T \end{cases} \tag{8a,b}$$

where *f(z)* is an arbitrary function describing the initial water content in the channel, Z is the lower boundary of the channel, and w_u is the water content in the channel at the upper boundary, *z=0*, during the square pulse input. The boundary condition when the water enters the channel only laterally is

$$w(z,t) = 0, \qquad z = z_A, \qquad 0 \le t \le T \tag{9}$$

where T is the duration of the drainage of water from the matrix into the channel domain. The initial condition given by (7) is also valid in this case.

Solution for case I

The solution for this case have been presented by Germann [1985]. However, for the sake of completeness, and for comparison with case II, the solution is derived here. The solution is derived under the initial condition given by (7), assuming that *f(z)=0*, and the boundary conditions defined in (8a,b). There is no lateral inflow and accordingly *r(z,t)=0*. The sudden increase in the water content at z=0 causes a discontinuity in the *w* profile, which will move downwards as a shock front, under the influence of gravity and maintain a sharp interface between the moisture content ahead and the moisture content immediately behind the shock front. The flow path of the shock is dictated by the basic conservation law itself and is determined by the flow behind and the flow ahead. In the case where the moisture content at the surface is suddenly decreased, a drainage front will develop which will travel at the speed defined by (4). The solution is divided into an infiltration domain and a domain of internal drainage, *D1* and *D2*, where the characteristic *z(t,T)* serves as a dividing line. The solution includes two free boundaries, the shock front *0-A* and *A-B*, which must be determined together with the solution. The characteristics carry a constant water content and therefore become straight lines. The solution domains are shown in Figure 3. Domain *D1* is bounded by the free boundary *0-A*, the drainage front (*z=z(t,T)*), and the t-axis. Domain *D2* is bounded by the drainage front, the free boundary *A-B*, the t-axis, and the lower boundary of the channel domain.

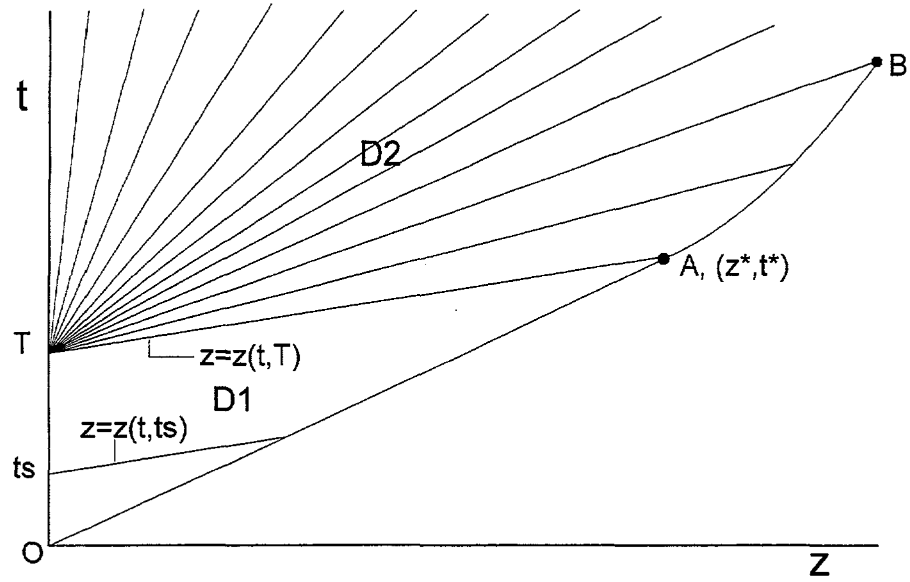

Fig.3 Characteristic diagram in case I.

The solutions of (5) and (6), subject to the initial and boundary conditions, become in *D1*,

$$w = w_u \tag{10}$$

$$z(t,t_s) = abw_u^{a-1}(t - t_s) \tag{11}$$

where t_s is the point at the t-axis at which the characteristic originates. Inserting $t_s=T$ into (11) yields the travel path of the drainage front.

In the domain of internal drainage, *D2*, the characteristics originate at time, T, and carry a water content which varies from one characteristic to the other, from w_u to zero. The characteristics form a fan shape and cover the whole domain *D2*. The solution becomes

$$w = w_s \tag{12}$$

$$z(t,w_s) = abw_s^{a-1}(t - T), \qquad t > T \tag{13}$$

where w_s is the water content which can take any value between w_u and zero.

Considering the law of conservation of mass, the velocity of the shock front can be written as [Smith,1983]:

$$\frac{dz}{dt} = \frac{q(w) - q(w^*)}{w - w^*} \tag{14}$$

where w is the water content in the shock front and w^* is the water content ahead of the front. In this case we assume that w^* is zero and the time and space history of the shock front becomes

$$z = bw^{a-1}t \tag{15}$$

from (11) and (15) it can be seen that the drainage front travels a times faster than the shock front *0-A*. Combining (11) and (15) gives the intersection point (z^*,t^*):

$$(z^*, t^*) = \left(\frac{abw_u^{a-1}T}{a-1}, \frac{aT}{a-1} \right) \tag{16}$$

If the drainage front overcomes the shock front before the lower boundary is reached the free boundary *A-B*, which is the path of the shock front as the shock strength diminish, must be determined. If one considers the characteristic $z(t,w_s)$ in domain *D2* it will intersect the free boundary at the point $(\zeta(w_s), \eta(w_s))$. The parametric representation of the free boundary becomes $z=\zeta(w_s)$ and $t=\eta(w_s)$. Inserting these into (13) yields:

$$z(\eta(w_s), w_s) = \zeta(w_s) = abw_s^{a-1}(\eta(w_s) - T) \tag{17}$$

The velocity of the free boundary *A-B* is given by (14) and for an initially dry channel, $w^* = 0$ we get:

$$\frac{dz}{dt} = \frac{\zeta'(w_s)}{\eta'(w_s)} = bw_s^{a-1} \tag{18}$$

Differentiating (18) yields:

$$\zeta'(w_s) = ab\Big((a-1)w_s^{a-2}(\eta(w_s) - T) + w_s^{a-1}\eta'(w_s)\Big) \tag{19}$$

By eliminating $\zeta'(w_s)$ in between (18) and (19) we get

$$\eta'(w_s) + \eta(w_s)\frac{a}{w_s} = T\frac{a}{w_s} \tag{20}$$

Solving (20) subject to the initial condition in (16) yields:

$$\eta(w_s) = T\left(1 + \left(\frac{w_u}{w_s}\right)^a \frac{1}{a-1}\right) \tag{21}$$

Eliminating w_s between (17) and (21) gives the time and space history of the free boundary *A-B*:

$$z(t) = abw_u^{a-1}\left(\frac{T}{a-1}\right)^{\frac{a-1}{a}} (t-T)^{\frac{1}{a}}, \qquad t \geq t^* \tag{22}$$

This completes the solution for case I.

Solution for case II

The solution is here derived for the case when water enters the channel domain laterally from a matrix segment, defined by $[z_A, z_B]$, at a steady rate r during the time interval T. The characteristic that originate in the point $(z_A, 0)$ at the z axis is the bounding characteristic and the time point where it intersects the line $z=z_B$ is the equilibrium time, t_{eq}. This is the time taken for a water molecule that enters laterally at $z=z_A$ to reach $z=z_B$. It is here assumed this time is shorter than the duration of a period of lateral inflow from the matrix. The solution is therefore derived for the case when $t_{eq} \leq T$ under the boundary condition (9). The parameter r is assumed to be constant in time and space and the initial condition $f(z)$ is, for simplicity, set to zero.

The solution can be divided into five domains, *D1*, *D2*, *D3*, *D4*, and *D5*. When the lateral inflow starts, a downward flow will immediately develop. This is the domain *D1* which is defined by the z-axis, the line $z=z_B$ and the bounding characteristic $t=(z,z_A)$. The flow will be unsteady and uniform, whereas the flow in domain *D2* is steady and uniform. Domain *D2* is bounded by the bounding characteristic $t=(z,z_A)$, $z=z_A$, $z=z_B$, *and* $t=T$. The characteristics that originate on the z-axis in *D1* extends into domain *D3* and become straight lines since no lateral inflow takes place here. The front $(z_B,0)$-(z_C,t_C), $z=z_B$, and the characteristic (z_B,t_{eq})-(z_C,t_C) define domain *D3*. The flow here and also in domain *D5* is unsteady and nonuniform. In domain *D3* the characteristics that carries a higher water content and travel faster will overcome slower ones, the front will therefore gradually become steeper and a shock front will start to evolve. The shock front will fully develop at point (z_C,t_C) where the front path intersect the bounding characteristic. In domain *D4*, which is defined by the characteristics (z_B,t_{eq})-(z_C,t_C) and (z_B,T)-(z_D,t_D), $z=z_B$, and the shock front, the water content is constant and the shock path becomes accordingly a straight line. In domain *D5* the shock strength is decreasing and the velocity of the shock is retarding. Domain *D5* is bounded by $z=z_A$, $t=T$, the characteristic (z_B,T)-(z_D,t_D), and the flow path of the attenuating shock front beyond (z_D,t_D). The solution domains and the characteristics are shown in Fig. 4.

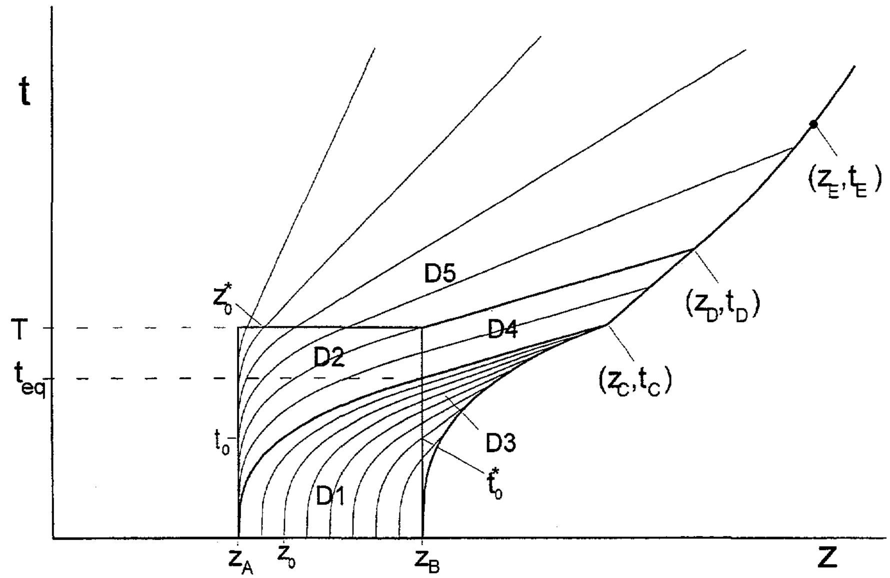

Fig.4 Characteristic diagram in case II.

The solutions of (5) and (6) under the initial and boundary conditions given by (7) and (9) is:

D1:

$$w = rt \tag{23}$$

$$z = z_0 + br^{a-1}t^a \tag{24}$$

D2:

$$w = r(t - t_0) \tag{25}$$

$$z = br^{a-1}(t - t_0)^a + z_A \tag{26}$$

where z_0 and t_0 are the points on the z axis and on the line z=z_A, respectively, at which the characteristics originate. Eliminating t_0 between (25) and (26) gives:

$$w = \left(\frac{(z - z_A)r}{b}\right)^{\frac{1}{a}} \tag{27}$$

This shows that the water content in domain *D2* is constant in time and only dependent on z.

The characteristics in domain *D3* originate from point t_0^* on the line z=z_B. They carry a constant water content which is defined along the boundary (z_B,0)-(z_B,t_{eq}) by (23). Inserting (23) into (6) and integrating yields the solution for the characteristics:

$$z = ab(rt_0^*)^{a-1}(t - t_0^*) + z_B \tag{28}$$

The parameter t_0^* may be eliminated between (23) and (28) to give an implicit expression for w as a function of z and t:

$$z = ab\left(w^{a-1}t - \frac{w^a}{r}\right) + z_B \tag{29}$$

In domain *D4* the water content is constant and is defined by (27) along the boundary (z_B,t_{eq})-(z_B,T):

$$w = \left(\frac{(z_B - z_A)r}{b}\right)^{\frac{1}{a}} \tag{30}$$

By inserting (30) into (6) and solving as previously we get:

$$z = ab^{\frac{1}{a}}\left(r(z_B - z_A)\right)^{\frac{a-1}{a}}\left(t - \left(\frac{z_B - z_A}{br^{a-1}}\right)^{\frac{1}{a}} - t_0\right) + z_B \tag{31}$$

In domain *D5* the boundary conditions can be written:

$$z(T) = z_0^* \tag{32}$$

$$w(z_0^*, T) = \left(\frac{(z_0^* - z_A)r}{b} \right)^{\frac{1}{a}}, \qquad z_A \le z_0^* \le z_B \tag{33}$$

where z_0^* is a parameter that represents the intersection of the line $t=T$ and the characteristics that originate in domain *D2*, $z=z(t,t_0)$. The characteristics in domain D5 carry a water content that is defined by the boundary condition (33). The water content along each characteristic is constant but differs from one characteristics to the other. By inserting (33) into (6) and solving subject to (32) we get

$$z(z_0^*, t) = ab^{\frac{1}{a}} (z_0^* r)^{\frac{a-1}{a}} (t - T) + z_0^* \tag{34}$$

It is apparent that the water content also in this domain is dependent on both z and t. Following the same procedure as previously, an implicit expression for w as a function of z and t is obtained by eliminating the parameter z_0^* between (33) and (34)

$$z(t, T) = b\left(aw^{a-1}(t - T) + \frac{w^a}{r} \right) \tag{35}$$

By inserting $z=\zeta(t_0^*)$ and $t=\eta(t_0^*)$ into (29) the free boundary (z_B,0)-(z_C,,t_C) can be represented in parametric form in terms of t_0^*. By following the earlier procedure the solution becomes:

$$z = br^{a-1} \left(\frac{at}{a+1} \right)^a + z_B \tag{36}$$

The intersection point (z_C,t_C) cannot be determined explicitly. Combining (28), with $t_0^* = t_{eq}$, and (36) the time coordinate, t_C, for the intersection can be expressed as:

$$t_C = \frac{t_{eq}^{1-a}}{a} \left(\frac{at_C}{a+1} \right)^a + t_{eq} \tag{37}$$

Equation (37) has to be solved by a iterative procedure. By inserting the result into (36) z_C is obtained.

The path of the shock front (z_C,t_C)-$(z_D$-$t_D)$ is yielded by inserting (30) into (14) and integrating:

$$z = b^{\frac{1}{a}}\left(r(z_B - z_A)\right)^{\frac{a-1}{a}}(t - t_C) + z_C \tag{38}$$

By eliminating z between (31) and (38) the time point when the boundary (z_B,T)-(z_D,t_D) and the shock front intersect is obtained:

$$t_D = \frac{aT - t_C}{a-1} \tag{39}$$

The time and space history of the attenuating shock front, beyond (z_D,t_D) can be derived in the same manner as previously. However, an explicit solution is not possible. A numerical solution is the only resort. The parametric representation of the attenuating shock path can be written:

$$z = \zeta(z_0^*) = a\left(\frac{br^{a-1}}{z_0^*}\right)^{\frac{1}{a}} z_B\left(t_D - T\right) + \frac{a^2}{a^2+1}\left(\left(\frac{{z_B}^{a+1}}{z_0^*}\right)^{\frac{1}{a}} - z_0^*\right) + z_0^* \tag{40}$$

$$t = \eta(z_0^*) = \left(\frac{z_B}{z_0^*}\right)\left(t_D - T\right) + \frac{a}{(a^2-1)\left(br^{a-1}\right)^{\frac{1}{a}}}\left(\frac{{z_B}^{\frac{a+1}{a}}}{z_0^*} - {z_0^*}^{-\frac{1}{a}}\right) + T \tag{41}$$

where

$$z_0^* \in \left[z_A, z_B\right]$$

The different stages of the propagation of a lateral inflow pulse, as described by (23)-(41), are illustrated in Fig.5 by the hydrograph at the depths z_B, z_C and z_E. The typical nonlinear behavior of the kinematic wave is clearly seen. As the wetting front is propagating downwards it steepens until a shock front is formed. This is due to that higher water contents travel faster than lower ones. On the back side, the same mechanism causes the wave to spread out.

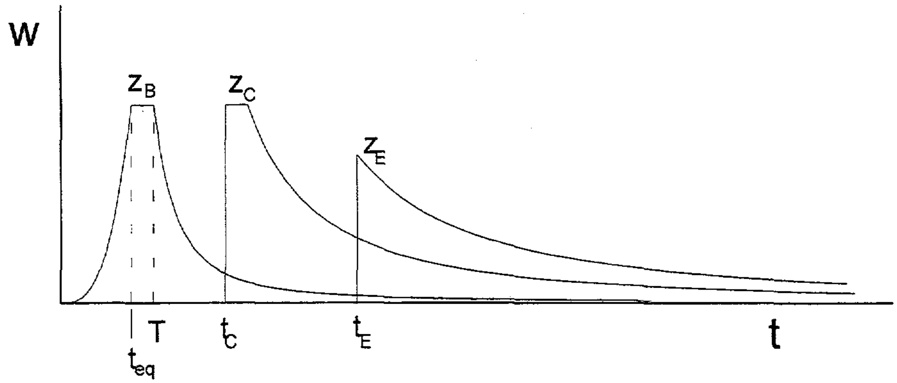

Fig.5 The resulting hydrograph from a steady lateral inflow r at $z_A \leq z \leq z_B$ with the duration T at the depths z_B, z_C and z_E.

Model applicability

Inspired by the comparison of the HELP model and PREFLOW model by Uguccioni and Zeiss [1997] we are proposing that the model for channel flow presented here may be coupled with the HELP model to make it capable of predicting fast responses to rainfall events and early breakthrough times. It is true that the HELP model has been extended with a layer designed to account for channel flow. However, since the effect of channel flow is that the bulk of the landfill is shortcut, a vertical channel domain, as in the PREFLOW model, may represent the channel phenomena better physically. The channel flow in the PREFLOW model is described by the Poiseuille's law, which constitute a special case of the more general power flux law employed here, may not be appropriate. According to our earlier discussion, flow in landfilled waste is more likely to take place as a thin film on solid surfaces than saturated flow in circular channels.

By lumping the channel network into a vertical one-dimensional channel domain and couple this with the HELP MSW layer [HELP layer #18, Schroeder et al., 1994] a dual domain model is obtained. The water flow in the matrix, can be calculated with the existing routing procedure. But instead of routing all excess water up to the previous segment, in case the storage capacity is filled in the actual segment, a portion can be re-routed into the channel domain. Water may also be re-routed to the channel domain if the water outflow rate of the previous segment exceeds the saturated hydraulic conductivity of the actual segment. In addition to lateral inflow from the matrix, the water may also flow into the channel domain through the upper end, that is the interface between the waste layer and the soil cover. Water that is flowing down the walls of the channels can also be sorbed into a matrix segment laterally by capillary forces.

SUMMARY AND CONCLUSIONS

The prominent observed characteristics of landfills have not been fully addressed in the literature on landfill hydrology. The flow and transport models used at present rely entirely on concepts and laws derived from soil science. Richards equation has been employed to model the water flow and the CDE to describe the transport process. However, the underlying criterion on which the process assumptions are based, has not been justified.

It can be concluded that the internal geometry of the waste medium is highly heterogeneous and shows anisotropy in the horizontal and vertical directions. Spatial periodicity on three scales and temporal variability due to the biodegradation process can be identified. The flow field is accordingly non-uniform, as a result of locally saturated regions or flow through restricted channels. Channel flow is most significant for young landfills and can be observed as leachate production well ahead of the time at which the landfill reaches field capacity. This calls for an appropriate flow model, and the waste medium should not, as is commonly seen in the literature, be characterized as a homogeneous soil with special properties.

In this paper, it is proposed that the landfill should be conceptualized as a dual-domain medium, consisting of a matrix and a channel domain. Richards equation may be applicable in the matrix domain where flow is slow and can be assumed to be fairly uniform. The flow in the channel domain occurs as a slender viscous film on solid surfaces and is mainly gravitational. Based on this, a strictly convective power law expression was employed as the governing flux law which, in combination with the continuity equation, produces a kinematic wave equation. A kinematic wave framework has been outlined for channel flow and solutions given for two cases when water enters the channel from the upper end and when water enters laterally. The kinematic wave model is simple and allows the method of characteristics to be used, which provides analytical solutions in most cases. A drawback of the kinematic wave model is that it cannot describe dispersion due to the basic kinematic assumption which is introduced by applying a strictly convective flux law.

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Paper V

Kinematic Wave Model for Water Movement in Municipal Solid Waste

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Abstract The movement of water in a large (3.5 m^3) undisturbed sample of 22-year-old municipal solid waste has been modeled using a kinematic wave approximation for unsaturated infiltration and internal drainage. The model employs a two-parameter power expression as macroscopic flux law. The model parameters were determined and interpreted in terms of the internal geometry of the waste medium by fitting the model to one set of infiltration and drainage data. The model was validated using another set of data from a sequence of water input events. The results of the validation show that the model performs satisfactorily, but further development of the model to incorporate spatial variability would increase its capability.

INTRODUCTION

The concentration of substances such as heavy metals, nutrients and organic compounds in landfills produces mass and energy gradients within and between the landfill and its surrounding environment. As a result, these substances will leave the landfill with gas or water flow as long as the gradients remain. Landfills therefore constitute a serious environmental problem. The aim of modern landfill management is to level out the energy and concentration gradients between the landfill and the environment in a controlled and sanitary manner. The final product is stabilized waste which may become an integrated part of the natural environment. Since the presence of water in the landfill plays a key role in the biochemical processes [Augenstein and Pacey, 1991; Christensen and Kjeldsen, 1989; Ehrig, 1983; Klink and Ham, 1982; Leuschner and Melden, 1983; Straub and Lynch, 1982], the mapping of water flow and solute transport is of direct interest in engineering the stabilization processes and when developing models for leachate quality and gas production.

A modest number of flow and solute transport models developed for waste media on a laboratory scale have been published. In a review by Bendz et al. [1998] it is pointed out that these models all treat the waste as a homogeneous porous medium. It is explicitly or implicitly assumed that the experimental scale is large enough for a representative elementary volume (REV) to exist so that a macroscopic approach can be justified. The water movement has been modeled by Richards equation and the transport process, when included, by the classical convection-dispersion equation (CDE). Furthermore, the soil water diffusivity, defined by Klute [1952], and the forms of hydraulic conductivity and capillary potential suggested by Clapp and Hornberger [1978] have been employed. However, as pointed out by Bendz et al [1998], the process assumptions on which the Richards equation and the ensuing CDE rely for their applicability have not been discussed. Straub and Lynch [1982] approached the transport of inorganic contaminants in landfills with a simple well-mixed reactor model and a continuum flow model and applied them to experimental data obtained in the laboratory by others. Influenced by the work of Straub and Lynch [1982], Demetracopoulus [1986] used the same mathematical framework but employed another numerical scheme to solve the differential transport equations. Korfiatis et al. [1984], who investigated water flow on a column scale, and Straub and Lynch [1982] found that the capillary diffusivity was of minor importance and could be neglected when the moisture content exceeded the field capacity. Vincent et al. [1991] used an artificial model waste and Lee et al. [1991] used shredded municipal solid waste of four different ages when performing laboratory experiments to investigate the flow and transport processes.

Straub and Lynch [1982] considered the heterogeneity and channeling effects on an ad hoc basis by simply adjusting the dispersivity parameter in the transport equation. Molecular diffusion was regarded as negligible. In the same way, Korfiates [1984] suggested that the channeling could be taken into account by increasing the exponent in the Clapp and Hornberger [1978] expression for the tension head.

An alternative approach to describing the flow in landfills, taking the geometrical configuration into account, was suggested by Ferguson [1993]. With the primary objective of estimating the specific surface in a landfill, he suggested a hydraulic model where water is present either as a static surface tension film or as a moving film on refuse particles.

Due to the highly heterogeneous nature of a landfill the flow field is not uniform. The internal geometry of a landfill facilitates fast flow in restricted channels and voids. Further, the field capacity is spatially variable and some parts of the landfill therefore reach field capacity long before the entire landfill does. Water may be flowing in locally saturated regions, while the largest portion of the landfill may be well below field capacity. Channel flow, which is most significant in young deposits due to their coarser structure, has been observed in several investigations [Bengtsson et al., 1994; Blakey, 1982; Blight et al., 1992; Harris, 1979; Ham and Bookter, 1982; Holmes, 1983; Korfiates et al., 1984; Walsh and Kinman, 1979]. Stegmann and Ehrig [1989] refer to investigations in which excavated landfills have shown evidence of channel flow. As the refuse biodegrades and settles, the dry density increases and the void volume decreases. This, in turn, limits the fast channel flow to some extent in older deposits. The extent of channel flow is not only dependent on the structure and void volume but also on the rate of precipitation. Additional flow routes may develop during periods of high infiltration rate [Jasper et al., 1985]. The existence of channel flow is very important and is believed to be the reason why existing models are not in agreement with actual field observations [Ehrig, 1983; Stegmann and Ehrig, 1989, Zeiss and Major, 1993]. The existence of channel flow also introduces additional complexity into the dispersion process as the transport process in these flow routes may be different from that within the rest of the medium.

To conclude, the common approach when addressing flow and transport in landfills has been to regard the waste medium as a homogeneous soil with special properties. The prominent phenomena of heterogeneity of the landfill and of channel flow have not been fully addressed in the existing models.

The objective of this paper is to present a framework for the modeling of unsteady water flow in a landfilled municipal solid waste considering the aforementioned characteristics of the landfill. In the model proposed here, the heterogeneity of the waste medium is addressed by discretizing the medium into a channel domain, constituted of the flow paths, and a matrix domain. The vertical water flow in the latter domain is assumed to be negligible. In the channel domain, the water is assumed to move as a creeping flow in thin layers driven by gravity on solid surfaces, and capillary forces are considered to be minor. The kinematic wave assumption is introduced by employing a strictly convective flux law. Several researchers have investigated the kinematic wave approximation for soil water movement [e.g. Sisson, 1980; Smith, 1983; Singh and Joseph, 1993]. The topic is discussed comprehensively by Singh [1997]. The kinematic wave approach to macropore flow in soils proposed by Beven and Germann in a series of papers is followed in this study [Beven and Germann, 1981; Germann, 1985; Germann and Beven, 1985, Germann, 1990; Germann and DiPietro, 1996].

KINEMATIC WAVE MODEL

Conceptualization of the medium and the flow regime

Bendz et al. [1998] identified the heterogeneity (variable in time and space), fast gravitational flow in restricted channels and voids, low capillarity, significant horizontal stratification (resulting from the disposal procedure) and impermeable surfaces as major features that govern the flow regime in landfills. The authors proposed that the spatial variability of the landfill may be categorized into three heterogeneity scales which were denoted the lens scale (lens-shaped refuse element), the truckload scale (a group of refuse lenses that derives from the same refuse truck load and may therefore share some characteristics), and the landfill scale. A lens-shaped refuse element is typically a compacted refuse bag/sack with a partlially impermeable surface. The lens scale is of the order of 0.5 m in the horizontal plane and 0.1m in the vertical plane.

The flow regime was represented as follows. Due to stratification, a significant portion of the flow is taking place in the horizontal direction. These flow paths at different levels are connected by vertical short-cuts. This leads to a network of flow paths, similar to the flow paths in fractured rocks or fissured media. An illustration of this flow pattern in two dimensions is shown in Fig.1.

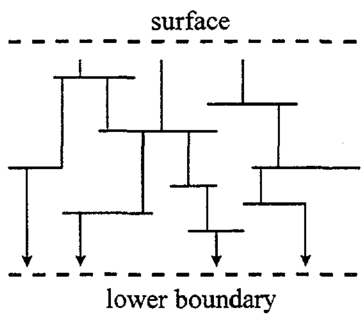

Fig.1 Flow paths in the landfill interior illustrated schematically.

Based on the flow pattern described above, the landfill is conceptualized as a dual-domain medium. The channel domain defined by the flow path network constitutes only a fraction of the entire landfill. Gravity is assumed to be dominant here and capillary force is considered to be negligible. In the matrix domain the capillary force is considered to be significant and the water movement is slow. In the model presented in this paper the vertical flow in the matrix domain has been neglected.

Governing flux law

In a critical review of Richards equation, Germann [1990] demonstrated macropore flow in soils to be one kind of deviation from the underlying assumption of perfect correlation between capillary diffusivity and capillary potential on the scale of a representative elementary volume (REV). That is, the water movement may be faster than the change in capillary potential. Bendz et al. [1998] acknowledged the similarities between channel flow in landfilled waste and macropore flow in soils and employed the two-parameter power function proposed by Beven and Germann [1981] as a macroscopic flux law,

$$q = bw^a \tag{1}$$

where q is the water flux density (ms^{-1}) in the channels per unit cross-sectional area of the medium, a is a dimensionless exponent and b is the channel conductance (ms^{-1}) which can be interpreted as the lumped effect of surface, geometrical, and spatial characteristics of the flow path. The amount of water in the channels is expressed as a volume fraction of the medium, w, rather than the thickness of a water film. Germann [1990] lent the flux law further credibility by treating the flow as a viscous laminar flow in thin layers and deriving a cubic form of equation (1) considering flow along a planar solid surface. Newton's law of shear stress was employed to that end. Such flow is often called creeping flow and is characterized as a moving fluid where the inertia is dynamically insignificant compared with the viscosity [Sherman, 1990]. Slender viscous films with free surfaces constitute a special case of creeping flows. The exponent a was reasoned by Germann and Di Pietro [1996] to be dependent on the internal geometry of the medium, the initial water content and the boundary conditions affecting the water content at the surface. The fluid-mechanical interpretation of the parameter a was summarized as a measure of the impact of the stagnant parts of a flow system on the mobile parts. For macropore flow, the parameter a was expected to be in the range of $2 \le a < \approx 8$, where a=2 indicates laminar flow in a cylindrical pipe (which may occur if the medium is saturated), and a=3 indicates flow in planar cracks. Turbulent flow is indicated when a<2. Higher values of a indicate increasingly tortuous flow. Completely dispersive flow corresponds to $a > \approx 10$.

It can be noted that flow in porous media shares certain characteristics with overland flow, such as gravity being the dominating force governing the water movement and viscosity being the only force in opposition [Singh, 1997]. Generally, surface flow can be expressed as

$$Q \propto h^a \tag{2}$$

where Q is the volume flux, h is water depth, and α is a dimensionless constant. The Chezy and Manning equations constitute special cases, where α equals $^3/_2$ and $^5/_3$, respectively.

Kinematic wave equations

The governing equations are the law of conservation of mass and a flux law. The conservation of mass equation can be expressed as

$$\frac{\partial w}{\partial t}+\frac{\partial q}{\partial z}=-S \tag{3}$$

where S is the water loss (s^{-1}) from the system of channels into adjacent pores and voids in the drainable region of the matrix domain, t is the time (s) and z is the depth (m). For a kinematic treatment we may assume that at a fix z, q is a function of w alone. Equation (3) can then be written as a kinematic wave equation:

$$\frac{\partial w}{\partial t}+c\frac{\partial w}{\partial z}=-S \tag{4}$$

in which

$$c=\frac{\partial q}{\partial w} \tag{5}$$

where c is the wave celerity (ms^{-1}). The average velocity with which w moves, u (ms^{-1}), and the wave celerity (ms^{-1}), follow from equation (1).

$$u=\frac{q}{w}=bw^{a-1} \tag{6}$$

$$c=\frac{\partial q}{\partial w}=abw^{a-1} \tag{7}$$

Substitution of equation (7) into equation (4) gives

$$\frac{\partial w}{\partial t}+abw^{a-1}\frac{\partial w}{\partial z}=-S \tag{8}$$

which can be reduced to the following system of characteristic equations.

$$\frac{dw}{dt} = -S \tag{9}$$

$$\frac{dz}{dt} = abw^{a-1} \tag{10}$$

For a square pulse input, q_u, starting at t=0 and ending at t=T, the following initial and boundary conditions can be assumed:

$$w(z,0) = f(z) \tag{11a}$$

$$w(0,t_s) = \begin{cases} 0, & t_s \leq 0,\ t_s \geq T \\ w_u, & 0 < t_s < T \end{cases} \tag{11b,c}$$

where f(z) is an arbitrary function describing the initial water content, w_u is the water content in the channels at the upper boundary, z=0, during the square pulse input and t_s is the point on the time-axis at which the characteristic defined by equation (10) starts.

EXPERIMENTAL SETUP AND DATA

The experimental setup, instrumentation, and the total number of experiments performed on the column have been thoroughly described by Rosqvist and Bendz [1998]. Only the part of the experimental setup that is of direct interest for this study will be discussed here. The data used in this study constitute only part of the data generated by the earlier experiment.

The waste sample was taken in 1995 from a 22-year-old test cell originally containing shredded household waste. The dimensions of the sample are 1.93 m in diameter and 1.20 m in height. The size of the sample is about double the lens scale in the horizontal plane and about ten times the lens scale in the vertical plane. The composition of the waste was characterized in only a basic way when the test cell was constructed. The composition is given in Table 1.

Table 1. *Original* composition of the waste in the test cell [Persson and Rylander, 1977].

Material	Composition (weight %)
sludge	35
Paper, Puitriscable, Glass	47
Textiles	6
Metal	6
Plastic	3
Wood, Timber	3

The waste was mixed with sludge, 35% by weight, at the time of disposal. The test cell was compacted to a dry density of 380 kgm^{-3}. With the objective of studying the presence and forms of heavy metals in an old deposit during the stabilization phase, Flyhammar et al. [1997] investigated the current composition of the waste in the test cell. It was found that the fraction of paper had decreased by more than 40 weight % and that the easily degradable materials were almost completely degraded. The geometrical forms found in the waste were also investigated and are shown in Table 2. According to our observations, the waste was highly compacted and tightly clustered.

Table 2 Geometrical forms found in the waste [Flyhammar et al. 1997]

Geometrical forms	Size (cm)	wt % dry waste	Waste fraction
sheets and threads *	> 0.2 - 10^1	12.7	non-rigid plastic,textile leather, rubber
irregular structures†	> 0.2 - 10^1	28.1	rigid plastic, metal inert (glass, stones) animal (bones) vegetable (incl. wood)
cellulose fibres	> 0.2 - 10^0	39.9	paper
fine residuals	< 0.2	19.2	fines

* one and two dimensional structures
† three dimensional structures

The test cell was built in 1973 on a waste disposal site in an area with a very high groundwater table. Since the late 1970s, when the site was covered, closed and abandoned, the cell has been laying under the ground water table. The sample may therefore represent waste which can be expected to be found in the saturated well-degraded bottom layer of a landfill.

After the soil cover of the test cell had been removed a steel cylinder, measuring 1.93 m in diameter and 2 m in height, was carefully driven down 1.2 m into the waste by letting an excavator alternating between applying a pressure on the top of the cylinder and excavating the waste material around the cylinder. To facilitate the procedure, one man cut the waste with a sharp spade along the edge of the cylinder as it was pressed down. When the drainage layer of the test cell had been reached, a steel sheet was forced in under the cylinder and fixed by welding. The cylinder was then lifted up, weighed and brought to the laboratory where the bottom steel sheet was removed and the cylinder was installed on a stand. The stand was equipped with a drainage layer of coarse gravel and a drainage pipe. The column was sealed to the stand with silicone. The cylinder was equipped with an irrigation system of 19 microsprinklers evenly distributed on a circular bar which was constantly rotated to ensure that the water was applied evenly. The irrigation flux was measured with an electronic flowmeter and the outflow flux with a tipping bucket device connected to a data-logger.

The *current* dry density and field capacity were determined in the laboratory using a smaller undisturbed sample, measuring 300 mm in diameter and 440 mm in height. The dry density and the field capacity of the waste were found to be 590 kgm^{-3} and 0.41, respectively. The increase in the dry density, compared with the original value, can be attributed to biodegradation and settlement.

The effective porosity was determined by closing the outflow valve of the drainage pipe and saturating the column sample with water, and then measuring the water volume until the sample was totally saturated. When saturating the sample with water there seemed to be voids that were filled immediately and others that required up to 6 hours to fill. The former voids were roughly assumed to constitute the channel volume and are denoted channel porosity in Table 3. By neglecting the presence of air-filled voids the total porosity was calculated as the sum of the effective porosity and field capacity. The column sample was assumed to be at field capacity after allowing it to drain until no outflow was registered.

Table 3 Measured properties of the column sample.

Property	
Height (m)	1.20
Dry Density (kg/m^3)	590
Porosity (m^3/m^3)	0.53
Effective Porosity (m^3/m^3)	0.12
Channel Porosity (m^3/m^3)	0.093
Field Capacity (m^3/m^3)	0.41

The data from two of the experiments performed with the column setup were used in the present study. In experiment I, the time required for the infiltration front to reach the lower boundary of the column was measured when a steady flow of $2.87 \cdot 10^{-5}$ m^3s^{-1}, corresponding to a flux density of $9.80 \cdot 10^{-6}$ ms^{-1}, was applied. The water input was then stopped and the recession was recorded. These data were used to calibrate the model. In experiment II, a sequence of square pulses with a water flux of $1.14 \cdot 10^{-4}$ m^3s^{-1}, which corresponds to a flux density of $3.92 \cdot 10^{-5}$ ms^{-1}, and duration 6 min was applied beginning every 24th minute. The time-averaged flow during experiment II was set to be the same as the flow in experiment I. Data from experiment II were used for validation of the model.

KINEMATIC WAVE SOLUTIONS

The solution is derived for the case when a sequence of square pulses of volume flux density, q_u, with a duration T is supplied to the surface of the waste media. The applied water is assumed to rapidly find its way to the channel domain where it flows downwards. The solution is divided in two main domains, the infiltration domain and the drainage domain, denoted *D1* and *D2*, respectively. A portion of the drainage domain forms a third solution domain, *D3*. The solution domains are shown in Fig. 2.

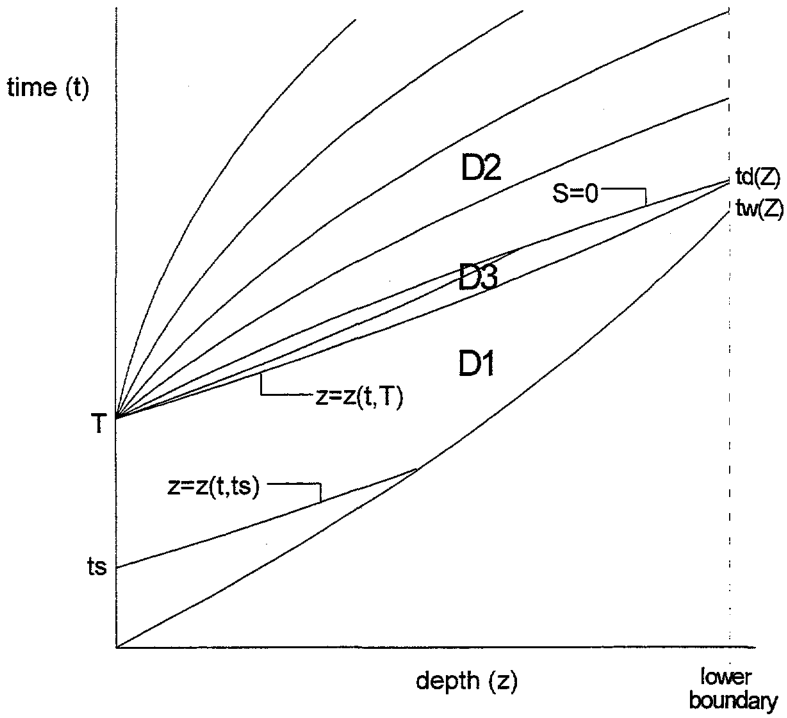

Fig.2 Solution domains for one wetting event.

Even though the column sample is at field capacity, the *S* term is not negligible in experiment II, which was used for validation, because the water flux during the pulses is high enough to force the water into voids and pores due to the potential gradient that builds up. When infiltration ceases, the water travels back to the channel system where it propagates downwards. The term *S* plays the role of an exchange term and switches signs depending on the direction of the hydraulic gradient between the matrix and the channel domain. In the infiltration domain, *D1*, *S* is positive. The characteristics originate from the t-axis on the segment $0 \le t \le T$. The position on the t-axis where the characteristic originates is t_s, and t is the parameter along the characteristics. The characteristic that originates from position T on the t-axis and divides domains *D1* and *D3* is the drainage front. In domain *D2*, water is flowing back into the channel system and the *S* term becomes negative. Domain *D3* is a region of internal drainage, however, the water potential is higher than in the storage region so the *S* term is still positive. The characteristic originating from the position T on the t-axis, which divides domains *D2* and *D3*, carries the water content at which the mobile and immobile regions are in equilibrium, that is, *S* is zero.

The characteristic solution in *D1* is given by solving equations (9) and (10) under boundary condition of equation (11c), and assuming that S is constant.

$$w(t,t_s) = w_u - S(t-t_s)\,, \qquad 0<t_s<T \tag{12}$$

$$z(t,t_s) = \frac{b}{S}(w_u{}^a - (w_u - S(t-t_s))^a) \tag{13}$$

By combining equations (12) and (13), t and t_s can be eliminated. Solving the resulting equation for $w(t,t_s)$ and inserting into equation (1) gives the following flux expression:

$$q\ (z,t) = b(w_u^a - \frac{zS}{b}) \tag{14}$$

This shows that the flux in domain *D1* is only dependent on z.
By rearranging equation (14), S can be determined as:

$$S = \frac{q_u - q_Z}{Z} \tag{15}$$

where q_u is the applied pulse flux density at z=0 and q_Z is the outflow flux density at depth z=Z.

The characteristic solution for domain *D3* involves a moving boundary along which the equilibrium between the mobile and immobile domains exists, so no interaction takes place. The equilibrium is assumed to appear at a certain water content in the channels, denoted by w_{eq}. The moving boundary gives the time-space history of S as it switches from positive to negative and water starts to flow back into the channels. Equations (12) and (13) become the same here as in *D1* except that the characteristics originate from the t-axis at the point T with the water content, w_s, which varies from one characteristic to the other, from w_u to w_{eq}.

The moving boundary is derived by inserting

$$w_{eq} = w_s - S(t-T) \tag{16}$$

into equation (13). The solution for the moving boundary then becomes:

$$w\ (t,T) = w_{eq} \tag{17}$$

$$z(t,T) = \frac{b}{S}((w_{eq} + S(t-T))^a - (w_{eq})^a) \tag{18}$$

In domain *D2*, the storage differs from the *S* term in *D1* and *D3*, not only by a shifted sign but also in magnitude, since water is forced into the smaller pores and voids during infiltration, whereas during drainage the flow of water back into the channel domain is governed by a much smaller hydraulic gradient. The flow back into the channels is assumed to be governed by the drainable water volume in the matrix, which has been accumulating in domain *D1* and *D3*. A variable function is therefore suggested, and equation (9) becomes:

$$\frac{dw}{dt} = kw_{mat} \tag{19}$$

where w is, as previously, the water content in the channels, k is a mass transfer rate coefficient (s^{-1}) and w_{mat} is the drainable volumetric water content in the matrix, stored in domains *D1* and *D3*. The time history of this water content in domain *D2* is defined by

$$\frac{dw_{mat}}{dt} = -kw_{mat} \tag{20}$$

Solving equation (20) under the initial condition $w_{mat}(t_{eq}) = \hat{w}_{mat}$, where t_{eq} is the time along the moving boundary, defined by equation (18), and $\hat{w}_{mat}$ is the maximum water content which has been stored, gives:

$$w_{mat} = \hat{w}_{mat} e^{-k(t-t_{eq})} \tag{21}$$

Inserting equation (21) into equation (19) and solving yields:

$$w = w_s + \hat{w}_{mat}(1 - e^{-k(t-t_{eq})}) \tag{22}$$

By combining equations (10) and (22) under the boundary condition (11d) the space and time history of the characteristics in domain *D2* becomes:

$$\frac{dz}{dt} = ab(w_s + \hat{w}_{mat}(1 - e^{-k(t-t_{eq})}))^{a-1} \tag{23}$$

As shown in Fig. 2, the characteristics originate from the position T on the t-axis and carry the water content defined by equation (22), where w_s varies from w_{eq} to zero. Thus, the characteristics intersect the t-axis at increasing angles.

The complete model must also include determination of the wetting front OA in domain *D1*. Smith [1983] formulated the shock velocity in the case where w decreases with depth as:

$$\frac{dz}{dt} = \frac{q(w) - q(w^*)}{w - w^*} \tag{24}$$

where w is the water content that propagates down along its characteristic defined by equation (12) and w^* is the water content in the waste medium ahead of the wetting front.

ESTIMATION OF PARAMETERS *a* AND *b*

For the special case when S is zero, as in experiment I, the solution domains reduce to *D1* and *D2*, as shown in Fig. 3. The characteristics carry a constant water content and therefore become straight lines. The solutions constitute special cases of equations (12), (13), (22), and (23). The solution in domain *D1* becomes:

$$w = w_u \tag{25}$$

$$z(t, t_s) = abw_u^{a-1}(t - t_s) \tag{26}$$

and in domain *D2* it becomes

$$w = w_s \tag{27}$$

$$z(t, T) = abw_s^{a-1}(t - T), \qquad t > T \tag{28}$$

where w_s varies from w_u to zero.
With $w^*=0$ the wetting front can be determined from equation (24) as:

$$z(t, t_s) = bw_u^{a-1}t \tag{29}$$

The drainage front results when $t_s = T$ is inserted into equation (26).

From equations (26) and (29) it can be seen that the drainage front travels a times faster than the wetting front.

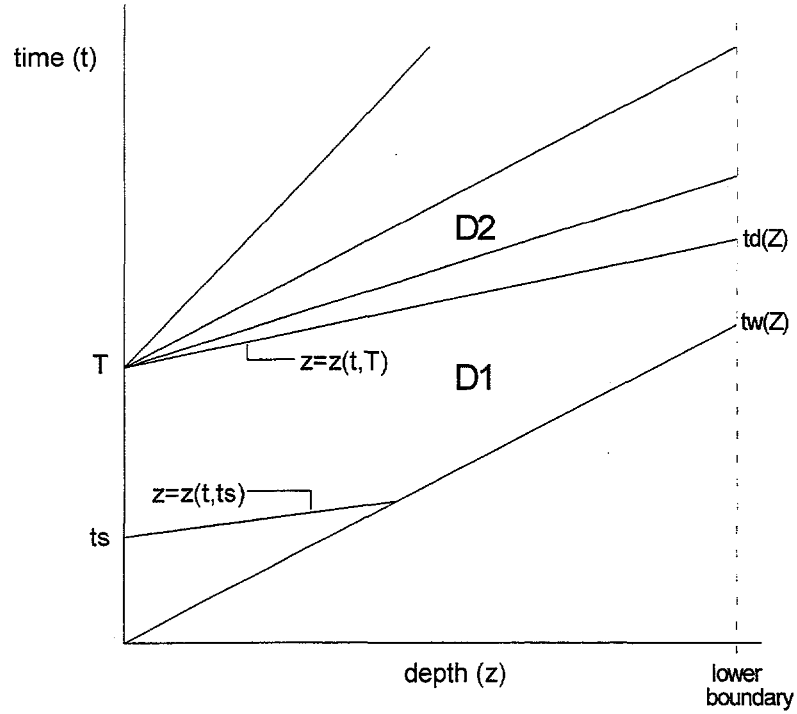

Fig. 3 Solution domains for a water input event when S is zero.

Following the method of Germann [1985], the flow parameters can be determined analytically using the data on the arrival time of the wetting front, the arrival time of the drainage front, and the recession hydrograph for a square pulse water input, in the case of S=0 ,as follows:

Combining equations (1) and (29), eliminating w and solving for b gives:

$$b = \left(\frac{Z}{t_w(Z)}\right)^a q_u^{1-a} \tag{30}$$

where $t_w(Z)$ is the arrival time of the wetting front at depth Z. By rearranging equation (28), the water content in domain $D2$ at depth Z and time t can be written as:

$$w = Z^{\frac{1}{a-1}}(ab(t-T))^{\frac{1}{1-a}} \tag{31}$$

Combining equations (1), (30) and (31) *q(Z,t)* in domain *D2* becomes

$$q = q_u\left(\frac{t_w(Z)}{a(t-T)}\right)^{\frac{a}{a-1}} \tag{32}$$

Equation (32) can be transformed into a linear equation by taking the logarithm:

$$q^* = \kappa t^* - \upsilon \tag{33a}$$

where

$$q^* = \ln\left(\frac{q(Z,t)}{q_u}\right) \tag{33b}$$

$$\kappa = \frac{a}{a-1} \tag{33c}$$

$$t^* = \ln\left(\frac{t_w(Z)}{t-T}\right) \tag{33d}$$

$$\upsilon = \frac{a}{a-1}\ln(a) \tag{33e}$$

Solving equation (26) for t gives the arrival time of the drainage front, $t_d(Z)$. The parameters u and v can be determined by fitting equation (33a) to the recession data using linear regression. Combining equations (33c) and (33e) the parameter a is given as:

$$a = e^{\frac{\upsilon}{\kappa}} \tag{34}$$

Inserting the value of a into equation (30) yields b.

RESULTS

Calibration

The data from experiment I were used to determine the model parameters, a and b, i.e. to calibrate the model. The waste was at field capacity and accumulation of water did not have to be accounted for (S=0). The arrival time of the wetting front at the outflow of the column was 1620 seconds. The solution for the calibration experiment is given by equations (25)-(29). The exponent a was determined by fitting equation (33a) to the recession data using linear regression and by employing equation (34). The result of the fitting procedure to the recession data is shown in Fig. 4. The coefficient of determination for the linear regression, r^2,was calculated to be 0.94.

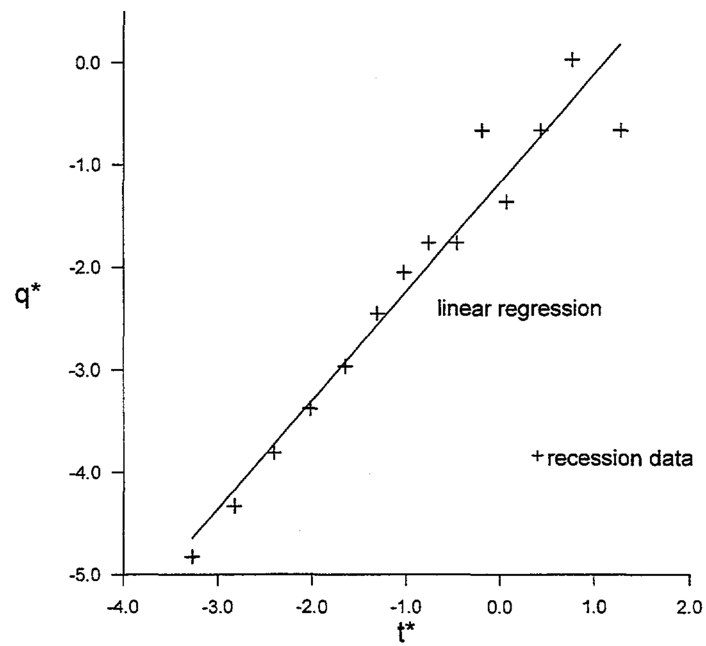

Fig.4 Fitting equation (33a) by linear regression to the recession data from experiment I.

The experimental recession curve is plotted in Fig. 5 together with the calibrated model. No data on the rising part of the hydrograph are available, as only the arrival of the infiltration front at the outflow was recorded. The arrival was defined as the point in time at which the outflow rate was equal to the input rate.

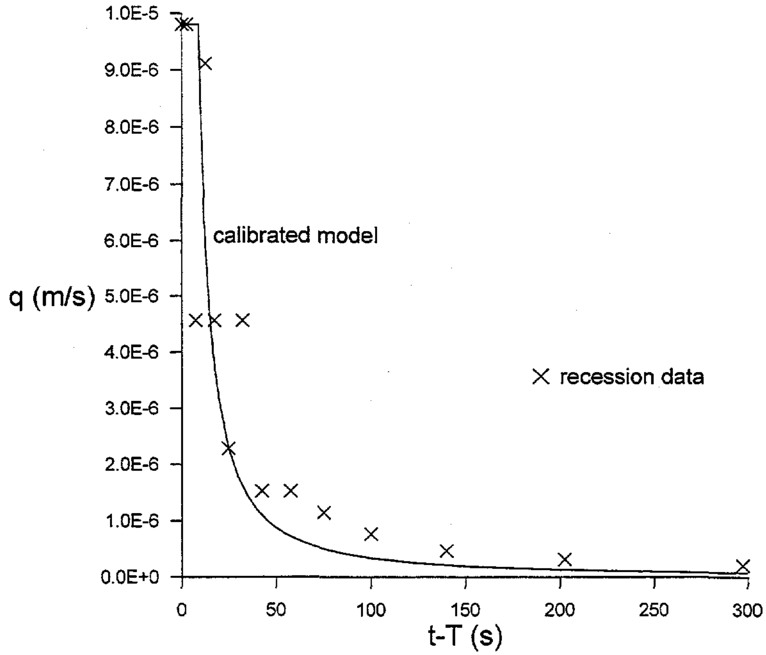

Fig.5 Experimental recession curve and calibrated model.

Result of the calibration:

$$\begin{cases} a = 3.05 \\ b = 5.24 \ ms^{-1} \end{cases}$$

According to the interpretation of the parameter a by Germann and Di Pietro [1996] given earlier in this paper, the value of a suggests that the flow can be characterized as flow along a planar surface. This interpretation agrees with the aforementioned void structure, involving horizontal stratification and the presence of impermeable surfaces.

Validation

The S term in domains $D1$ and $D3$ was determined using equation (15). The maximum water content that had been stored was determined from the recession curve after irrigation had been shut off. Given $\hat{w}_{mat}$ and the time it took to drain this volume, the mass transfer rate coefficient k was determined using equation (21).

The time-space history of the infiltration front was calculated by inserting q^* and w^*, as defined by equations (22) and (23), and q and w, as defined by equations (12) and (14), into equation (24). The final equation was solved by employing a simple numerical scheme. The outflow flux density in domain $D1$ was calculated using equation (14), whereas a numerical procedure was the only resort when calculating the outflow flux density in domain $D2$, by combining equations (1), (22) and (23).

The storage parameters were found to be:

$$\begin{cases} S = 2.11 \cdot 10^{-5},\ (s^{-1})\ (D_1\, and\, D_3) \\ \hat{w}_{mat} = 9.97 \cdot 10^{-3},\ (m^3 / m^3) \\ k = 1.0 \cdot 10^{-3},\ (s^{-1})\ \ (D_2) \end{cases}$$

The S term represents a flow of about $7 \cdot 10^{-5}$ m^3s^{-1}, in $D1$ and $D3$. By comparing this value with the applied water flux in the pulses of $1.14 \cdot 10^{-4}$ m^3s^{-1} and the maximum outflow flux of $4.0 \cdot 10^{-5}$ m^3s^{-1} it can be concluded that the water volume applied during the pulses splits approximately equally into flow from the channels into adjacent voids and pores, which may be assumed to be parallel to the horizontal stratification, and vertical flow.

The result obtained with the model is compared with experimental data in Fig. 6. The model predicts a net accumulation of about 5% of the applied water during the sequence of pulses. The model predicts this volume to be drained when the water input is shut off and the waste medium is allowed to drain for a longer period of time.

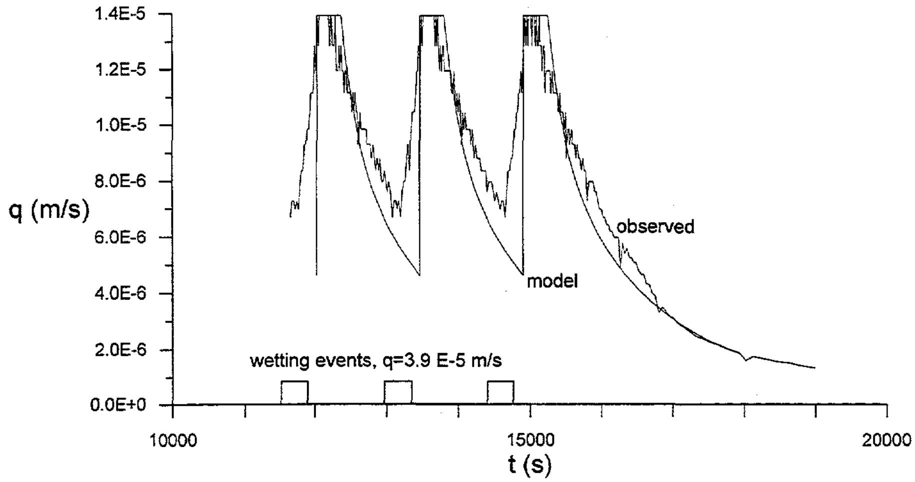

Fig. 6 The result obtained with the model compared with data from the column experiment.

DISCUSSION AND CONCLUSIONS

The kinematic wave model describes the arrival of the wetting front and the drainage front during unsteady flow in the investigated sample fairly accurately. However, the model is not capable of describing the observed dispersion, due to the basic assumption of the kinematic wave model and the flux law employed here, which is strictly convective. There is also a spatial variability in the waste medium, which is not taken into account, and this results in deviations from the modeled hydrograph. Water may infiltrate faster in some channels than the average rate, which can be seen as an earlier rise in the measured hydrograph compared with the model. It is also shown that the flow pattern is dependent on the applied flux density. In the square pulse experiment, the horizontal water flow from the channels into adjacent voids and fissures is significant compared with the average vertical flow.

The interpretation of $a=3.05$ suggests that a limited number of channels is present in which water movement takes place as a creeping flow in thin layers with free surfaces along planar solid surfaces.

Further development of the model, including the incorporation of spatial variability, in order to make it applicable on a larger scale, is planned.

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Paper VI

Solute Transport under Steady and Transient Conditions in Municipal Solid Waste

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Abstract The transport of a conservative tracer (lithium) in a large-scale (3.5 m^3) undisturbed municipal solid waste (MSW) sample has been investigated under steady and fully transient conditions using a simple model. The model comprises a kinematic wave approximation for water movement, presented in a previous paper, and a strict convective solute flux law. The waste medium is conceptualized as a three-domain system consisting of a mobile domain, an immobile active domain, and an immobile passive domain. The solute is assumed to travel in the mobile domain with a piston-type displacement. Diffusional mass exchange between the mobile and the active immobile water volumes is a reversible process and is assumed to be a linear function of the concentration gradient. The model has been used to interpret and compare the results from a steady-state experiment and an unsteady-state experiment. By solely adjusting the size of the immobile active fraction the model is capable of accurately reproducing the measured outflow chemographs for both the steady- and unsteady-state experiments. During unsteady conditions, the immobile active domain is found be about 65% larger than that under steady state conditions. It is therefore concluded that the water input pattern governs the size of the immobile active domain which, in turn, governs the solute residence time in the solid waste.

INTRODUCTION

Our understanding of the conditions and the biochemical processes which govern emission fluxes and biodegradation in the landfill interior is less than complete, and several researchers have emphasized the need for further research in the area. Straub and Lynch [1982] identified water movement, leachate strength and contaminant transport in the landfill interior as fields which require more attention. In modeling methane production, Augenstein and Pacey [1991] stated that the most important research task, in order to improve the performance of degradation models, is to investigate the presence and mobility of water.

Water constitutes the basic carrier for substances within a landfill. A high water content is therefore probably the most important prerequisite for anaerobic degradation processes [Ehrig,1991], as it facilitates the redistribution of nutrients and microorganisms within the landfill [Augenstein and Pacey, 1991; Christensen and Kjeldsen, 1989] while hydrolysis may be rate-limiting for the whole degradation process [Leuschner and Melden, 1983]. Also, the water flux has been shown to have a positive effect on the intensity of the biodegradation process [Klink and Ham, 1982]. In a review [Bendz et al., 1998a] of the published models, we have previously found that the prevailing approach for modeling of transport in a solid waste medium is to apply the Richards equation to determine the space-time history of the water velocity and to incorporate this into the convection-dispersion equation (CDE) to give the solute concentration [Demetracopoulus, 1986; Korfiatis et al., 1984; Lee et al., 1991; Straub and Lynch, 1982; Vincent et al., 1991]. This approach can be questioned since the basic assumption of the CDE, which justifies the formulation of the hydrodynamic dispersion flux as a diffusional flux, may not be fulfilled due to the considerable heterogeneity of the waste medium [Bendz et al., 1998a]. Further, the presence of fast channel, or macropore, flow constitutes a deviation from Richards' assumption of a perfect correlation between capillary diffusivity and capillary potential [Germann, 1990]. Solute transport under transient conditions is a complicated process. In addition to the hydraulic properties of the medium, Brusseau and Rao [1990] identified initial conditions and water input pattern at the upper boundary as factors which govern the transport process.

In Bendz et al. [1998a] we suggested that the landfilled waste medium can be discretisized into a matrix domain and a channel domain, which would allow separate assumptions regarding the flow and transport processes to be made in each domain. The channel domain was assumed to consist of a network of flow paths in which the movement of water is dominated by gravity. In the same study, a kinematic wave model for modeling channel flow was outlined and solutions were derived for three cases: water entering the channel from the end, water entering laterally through the boundaries of the channel and, finally, water entering the channel from both the end and laterally. We have previously applied the kinematic wave model to model the infiltration and drainage of a sequence of wetting events in a large-scale undisturbed MSW sample [Bendz et al., 1998b].

In this study, the model previously presented [Bendz et al., 1998b] was coupled to a piston flux law for solute transport to produce the space and time history of solute concentration. The mobile-immobile domain conceptualization (MIM) for solute transport as proposed by Van Genuchten and Wierenga [1976] was employed. The MIM model was further discretized by dividing the immobile domain into two sub-domains. Multi-domain models have been widely used to describe both transport- and sorption nonequilibrium, see the reviews by Brusseau and Rao [1990] and Sardin et al. [1991]. The MIM concept is useful in describing the effects of stagnant regions such as the tailing phenomenon. However, when used as a simulation model, it has a certain weakness since its parameters cannot be measured independently, but must be determined by simultaneously fitting the parameters to the outflow chemograph [Jury et al., 1991; White, 1985]. The model was used here only to parameterize and interpret results by calibrating it with data from solute experiments performed under steady and unsteady conditions.

The objective of this study was to evaluate the model on a field column and to investigate differences in the estimates of the parameters for steady and unsteady conditions. The experimental setup and the unsteady flow conditions are identical to those described in Bendz [1998b].

THEORY

The waste medium is represented by a matrix domain, in which capillary forces are dominant and the vertical movement of water is assumed to be negligible, and a channel domain in which water moves under the influence of gravity. Consequently, an applied water pulse at the surface is assumed to instantaneously find its way to the channel domain where it flows downwards.

The matrix domain is composed of the solid phase, a region in which water can be held against gravity by capillary forces, and a drainable water region. The space which is not filled with water is occupied by air or biogas. The water in the capillary region of the matrix domain is assumed to be stagnant and will therefore be referred to as immobile. The maximum water content of the capillary region is identical to the field capacity. The vertical flow of water in the drainable region of the matrix is assumed to be negligible. Water in this region is assumed to move mainly in the horizontal direction until it intersects the channel domain where gravity dominates and the water rapidly flows downwards. There may be a reversible water exchange between the channel domain and the drainable region of the matrix domain governed by a hydraulic gradient.

Solute transport is assumed to be strictly convective and is described by a simple piston flux law. Although a solute can move in the matrix domain by diffusion this form of transport is assumed to negligible in comparison with the fast convective transport in the channel domain. Solute transport between the regions is due to diffusion. Since the channel domain is very small the access to a large part of the medium is constrained by diffusional transport. The medium is therefore in a state of nonequilibrium [Brusseau and Rao, 1990]. The immobile domain is divided into two sub-domains. The part of the immobile domain which is located around the channels in which the water flow takes place is assumed to be active in the reversible exchanging of solute. Solute also diffuses deeper into the matrix, were the solute is not so easily recovered on a short time basis. This defines the passive immobile domain.

The diffusional mass exchange rate is assumed to be linearly proportional to the difference in concentration between the domains.

The domains and regions described above are illustrated in Fig.1.

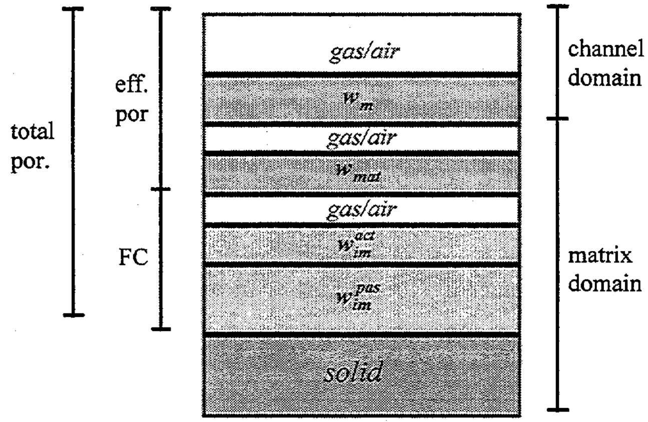

Fig. 1 Illustration of the different domains. *w* denotes the water content as a volumetric fraction of the total volume, the subscripts *m* and *im* denote the fractions of mobile and immobile water regions, respectively, the superscripts *act* and *pas* denote the active and passive parts of the immobile domain, *FC* is the field capacity, and w_{mat} is the drainable water content in the matrix. Note that the total porosity is assumed to be equal to the sum of the effective porosity and the field capacity. The non-interconnected closed pore volume is neglected.

Flow model

The space and time history of water content in the channel domain is given by the kinematic wave model presented in our earlier paper [Bendz et al., 1998b]. The framework of the flow model is summarized below. The flow in the channel domain is characterized as a thin viscous film with free surface. The water film moves, under the influence of gravity, along the surfaces that constitute the boundaries of the channels. Based on this, the following two-parameter power function, originally proposed by Beven and Germann [1981] for macropore flow in soils, was employed as a macroscopic flux law for channel flow in landfilled waste.

$$q = bw^a \tag{1}$$

where q is the water flux density (ms^{-1}) in the macropores per unit cross-sectional area of soil, a is a dimensionless exponent and b is the macropore conductance (ms^{-1}) which can be interpreted as the integrated effect of surface, geometrical, and spatial characteristics of the flow path. The amount of water in the macropores is expressed as a volume fraction of the soil, w, rather than the thickness of a water film.

We determined the parameters in the flux law, for the experimental setup that is used here, by a calibration procedure and found them to be [Bendz et al., 1998b]:

$$\begin{cases} a = 3.05 \\ b = 5.24 \quad ms^{-1} \end{cases}$$

The fluid-mechanical interpretation of the parameter a was summarized by Germann and Di Pietro [1996] as a measure of the impact of the stagnant parts of a flow system on the mobile parts. According to their fluid-mechanical interpretation of the magnitude of a, it was concluded that the channel flow in the waste medium could be characterized as flow along a planar surface, that is, flow along the boundaries of the channels.

The governing equations in the flow model are the law of conservation of mass and the flux law (1). The conservation of mass equation can be expressed as:

$$\frac{\partial w_m}{\partial t} + \frac{\partial q}{\partial z} = -S \tag{2}$$

where the subscript m denotes *mobile* and corresponds to the channel flow, S is the rate of water exchange between the channel domain and the matrix domain (s^{-1}), t is the time (s) and z is the depth (m).

Inserting (1) into (2) and, for a kinematic wave treatment, assuming that at a fixed z, q is a function of w alone yields a kinematic wave equation,

$$\frac{\partial w_m}{\partial t} + c\frac{\partial w_m}{\partial z} = -S \tag{3}$$

in which

$$c = \frac{\partial q}{\partial w_m} = abw_m^{a-1} \tag{4}$$

where c is the wave celerity (ms^{-1}). The last term follows from equation (1), and inserted into (3) gives:

$$\frac{dw_m}{\partial t} + abw_m^{a-1}\frac{\partial w_m}{\partial z} = -S \tag{5}$$

which can be solved analytically by the method of characteristics. Equation (5) is then reduced to the following system of characteristic equations:

$$\frac{dw_m}{dt} = -S \tag{6}$$

$$\frac{dz}{dt} = abw_m^{a-1} \tag{7}$$

For a square pulse input, q_u, at the surface, starting at t=0 and ending at t=T, the following initial and boundary conditions can be assumed:

$$w_m(z,0) = f(z) \tag{8}$$

$$w_m(0,t_s) = \begin{cases} 0, & t_s \le 0,\ t_s \ge T \\ w_u, & 0 < t_s < T \end{cases} \tag{9a,b}$$

where $f(z)$ is an arbitrary function describing the initial water content, w_u is the water content in the channels at the upper boundary, z=0, during the square pulse input and t_s is the point on the time axis where the characteristic defined by equation (7) starts

Given the initial and boundary conditions equation (6) and (7) can be solved. The solution is given in Bendz et al. [1998b] and will not be described here. The solution of (7) gives the time and space history of the characteristic along which the water content defined by the solution of (6) travels.

For a water input event at the surface the water content will increase abruptly and cause a discontinuity in the water content profile. This discontinuity will move downwards as a shock front maintaining a sharp interface between the water content of the front and the water content ahead.

By obeying the continuity equation, the shock velocity in the case where w decreases with depth can be written as [Smith,1983]:

$$\frac{dz}{dt} = \frac{q(w) - q(w^*)}{w - w^*} \tag{10}$$

where w is the water content, defined by (6) that propagates down along its characteristic defined by equation (7) and w^* is the water content in the waste medium ahead of the wetting front.

Transport model

The continuity equation for solute is written as

$$\frac{\partial wC}{\partial t} + \frac{\partial q_s}{\partial z} = -SC_m \tag{11}$$

where C is the total solute concentration (mg/l), w is the total water content expressed as a fraction of the total volume, q_s is the solute flux (m^3/m^{2}s) and SC_m represents the loss of solute due to convection into adjacent voids and pores in the matrix. Since the solute is assumed to be conservative, adsorption is not taken into account. Inspired by van Genuchten and Wierenga [1976] the total concentration is written as:

$$wC = w_m C_m + w_{im}^{act} C_{im}^{act} + w_{im}^{pas} C_{im}^{pas} \tag{12}$$

The immobile concentrations, C_{im}^{act} and C_{im}^{pas} are treated as constant in space so that there are no concentration gradients within the immobile domains. The mass transfer between the mobile and active immobile water regions is assumed to be reversible, diffusional, and linearly proportional to the concentration gradient. There is also diffusional transport from the active domain into the passive fraction of the immobile domain. This transport is assumed to be irreversible on a short time basis and, for simplicity, to be a linear function of the active immobile concentration only so that:

$$w_{im}^{pas} \frac{\partial C_{im}^{pas}}{\partial t} = \beta C_{im}^{act} \tag{13}$$

where β is the mass transfer rate coefficient (s^{-1}) for the diffusional mass flux between the active immobile domain and the passive immobile domain.

The diffusional mass transport between the mobile and active immobile domains is assumed to be governed by the concentration gradient and a mass transfer rate coefficient, α, (s^{-1}). The solute concentration in the active immobile domain can therefore be written as:

$$w_{im}^{act}\frac{\partial C_{im}^{act}}{\partial t}=\alpha(C_m-C_{im}^{act})-\beta C_{im}^{act} \tag{14}$$

The mass flux law of the commonly used CDE is

$$q_s=q_wC_m-D\frac{\partial C_m}{\partial z} \tag{15}$$

where q is the water flux (m/s) and D is the lumped dispersion-diffusion coefficient.
However, for simplicity, and since the basic assumption of the CDE could not be justified due to the heterogeneity of the medium, the dissipative term is neglected and the solute flux law becomes:

$$q_s=q_wC_m \tag{16}$$

By employing this flux law, the concentration front is assumed to move as a piston. The tailing phenomenon is represented solely by the mass transfer between the mobile and immobile regions.

Combining equations (11), (12) and (16) and assuming that the immobile water content is constant, yields:

$$w_m\frac{\partial C_m}{dt}+C_m\frac{\partial w_m}{\partial t}+w_{im}^{act}\frac{\partial C_{im}^{act}}{\partial t}+w_{im}^{pas}\frac{\partial C_{im}^{pas}}{\partial t}+q_w\frac{\partial C_m}{\partial z}+C_m\frac{\partial q_w}{\partial z}=-SC_m \tag{17}$$

Noting that the sum of the second term and the sixth term on the left-hand side equals the term on the right-hand side, equation (17) becomes:

$$w_m\frac{\partial C_m}{dt}+w_{im}^{act}\frac{\partial C_{im}^{act}}{\partial t}+w_{im}^{pas}\frac{\partial C_{im}^{pas}}{\partial t}+q_w\frac{\partial C_m}{\partial z}=0 \tag{18}$$

and inserting equations (13) and (14) into equation (18) gives:

$$\frac{\partial C_m}{\partial t}+\frac{q_w}{w_m}\frac{\partial C_m}{\partial z}=-\frac{\alpha}{w_m}(C_m-C_{im}^{act}) \tag{19}$$

Equation (19) is a linear first-order partial differential equation and can be solved using the method of characteristics. The characteristic equations are:

$$\frac{dC_m}{dt}=-\frac{\alpha}{w_m}(C_m-C_{im}^{act}) \tag{20}$$

and

$$\frac{dz}{dt}=\frac{q_w}{w_m}=u \tag{21}$$

According to equation (21), the concentration celerity is the same as the average water flow velocity.

The initial and boundary conditions are:

$$C_m(0,t)=C_u \qquad 0\leq t\leq T_p \tag{22a}$$

$$C_m(0,t)=0 \qquad t<0 \;\; and \;\; t>T_p \tag{22b}$$

and

$$C_{im}(z,0)=0 \tag{22c}$$

where C_u is the solute concentration in the applied pulse and $0\leq t\leq T_p$ is the time segment during which solute is applied at the surface.

Solutions

When the immobile concentration and the mobile water content are assumed to be constant in time, equation (20), under the boundary condition given by equation (22a), has the solution:

$$C_m(t,t_s)=C_{im}^{act}+(C_u-C_{im}^{act})e^{-\frac{\alpha}{w_m}(t-t_s)} \qquad 0\leq t_s\leq T_p \tag{23}$$

The solution shows that C_m declines exponentially along the characteristic defined by equation (21). The water content is constant only during steady-state flow, and the immobile concentration in the active region shows large temporal fluctuations and may therefore not be regarded as constant. Accordingly equation (23) is only applicable for short time intervals, Δt, and can then be written:

$$C_m(t+\Delta t,t_s) = C(t)_{im}^{act} + (C_m(t,t_s) - C(t)_{im}^{act})e^{-\frac{\alpha}{w_m(z,t)}\Delta t} \qquad 0 \le t_s \le T_p \tag{24}$$

The mobile water content at space-time coordinates (z,t), $w_m(z,t)$, is given by the flow model in Bendz et al. [1998b].

Substitution of equation (14) into equation (20) gives

$$\frac{dC_m}{dt} = -\frac{1}{w_m}\left[w_{im}^{act}\frac{\partial C_{im}^{act}}{\partial t} + \beta C_{im}^{act}\right] \tag{25}$$

The immobile concentration in the active region at time $t+\Delta t$ is calculated by employing a simple numerical scheme and solving equation (25) for $C_{im}^{act}(t+\Delta t)$:

$$C_{im}^{act}(t+\Delta t) = C_{im}^{act}(t) - \frac{1}{w_{im}^{act}}\left[\beta C_{im}^{act}(t)\Delta t + \frac{1}{Z}\sum_{t_s=t_s^{\min}}^{t_s=t_s^{\max}} \Delta z \cdot \left[C_m(t+\Delta t,t_s) - C_m(t,t_s)\right]\right] \tag{26}$$

where:

$$\Delta z = \frac{1}{4}\left(z(t+\Delta t,t_s-1) - z(t+\Delta t,t_s+1) + z(t,t_s-1) - z(t,t_s+1)\right)$$

$$\overline{w}_m = \frac{1}{2}\left(w_m(t+\Delta t,t_s) + w_m(t,t_s)\right)$$

$t_s^{\min}$ and $t_s^{\max}$ denote the time of origin of the "oldest" and the "newest" characteristics in the ensemble, respectively, and Z is the total depth of the sample .

The chemograph at the lower boundary, Z, can be determined by a routing procedure, using equations (24) and (26) under the appropriate initial and boundary conditions. For each time step, Δt, and under the condition that there is water input, P, a characteristic is added such that $t_s^{\max} = t + \Delta t$ and carries the boundary conditions given by equation (22a-b). The mobile solute concentration is calculated with equation (24) for the whole ensemble of solute characteristics for each time step. By inserting expressions for w(z,t), given in Bendz et al. [1998b], into equation (21), and employing a Runga-Kutta procedure, the z coordinate at the new time, $z(t+\Delta t,t_s)$, can be calculated for each solute characteristic. If a characteristic reaches the lower boundary, defined by $|z(t+\Delta t,t_s) - Z| < \xi$, where ξ denotes an arbitrary small number, $C_m(t+\Delta t,t_s)$ is registered as the outflow concentration, $C(Z,t+\Delta t)$. The characteristic is thereafter removed from the ensemble of characteristics so that the time of origin of the "oldest" characteristic, $t_s^{\min}$, becomes $t_s^{\min} + \Delta t$. The new immobile solute concentration is then calculated with equation (26). A schematic illustration of the procedure is given in Fig.2.

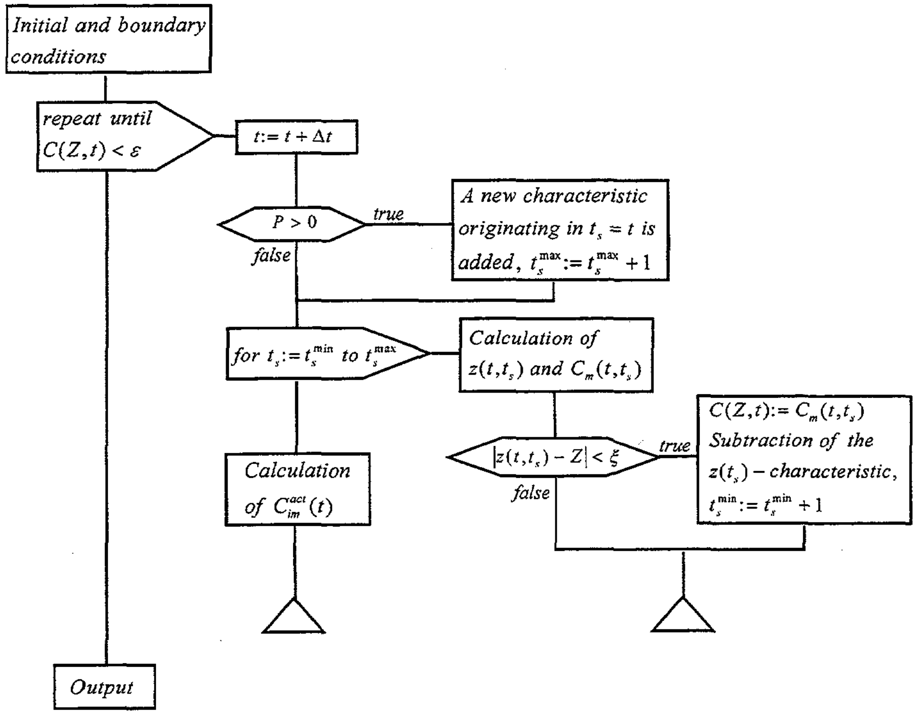

Fig.2 The routing procedure presented as a flow chart.

EXPERIMENTAL SETUP

Rosqvist and Bendz [1998] have in detail described the sampling procedure, experimental setup, and the total number of experiments performed on the column. Therefore, only the part of the experimental setup that is of direct interest in this study is described here.

The sample was taken in 1995 from a 22-year-old test cell, originally containing shredded household waste. The dimensions of the sample are 1.93 m in diameter and 1.20 m in height. The composition of the waste at the time of disposal is given in Table 1. The test cell was compacted with conventional methods to a dry density of 380 kgm^{-3}.

Table 1. *Original* composition of the waste in the test cell [Persson and Rylander, 1977].

Material	Composition (weight %)
sludge	35
Paper, Puitriscable, Glass	47
Textiles	6
Metal	6
Plastic	3
Wood, Timber	3

Since the late 1970s, the cell has been laying under the groundwater table.In a recent investigation of the current composition and geometrical forms of the waste, Flyhammar et al. [1997] found that the fraction of paper had decreased by more than 40% by weight and that the easily degradable materials were almost completely degraded.

The current properties of the sample is summarized in Table 2 [Bendz et al., 1998b]. The total porosity was determine by saturating the sample, i.e., the presence of closed air-filled voids was neglected.

Table 2 Measured properties of the column sample [Bendz et al., 1998b].

Property	
Height (m)	1.20
Dry Density (kg/m^3)	590
Porosity (m^3/m^3)	0.53
Effective Porosity (m^3/m^3)	0.12
Channel Porosity (m^3/m^3)	0.093
Field Capacity (m^3/m^3)	0.41

According to our observations, the waste was highly compacted, stratified, and tightly clustered. The increase in the dry density, compared to the original value, is an effect of biodegradation and settlement. The sample may represent highly compacted well-degraded waste, which can be expected to be found at larger depths in landfills.

With the aid of a excavator the sample were taken by a punching technique. A steel cylinder, measuring 1.93m in diameter, was pressed down into the waste. In this way a cylindrical volume was cut out. At the depth of 1.2 m a steel sheet was forced in under the cylinder and tack-welded in place. The cylinder was then lifted up and brought to the laboratory where the steel sheet was removed and the cylinder was installed on a stand equipped with a drainage layer and a drainage pipe. On top of the cylinder a irrigation system of 19 micro sprinklers were installed on a bar that was kept rotating to secure that the sample was irrigated evenly. The irrigation flux was measured with an electronic flowmeter and the outflow flux with a tipping bucket device connected to a data-logger.

The column sample was at field capacity at the beginning of the experiments.

EXPERIMENTS

Two solute experiments, under steady and unsteady conditions, are presented in this paper. Lithium bromide (LiBr) was used as a conservative tracer in both experiments. The same mass of LiBr was used in both experiments.

The first solute experiment was performed during steady unsaturated flow to calibrate the model parameters. The flow was kept steady at $2.87 \cdot 10^{-5}$ m^3s^{-1}, corresponding to a flux density of $9.80 \cdot 10^{-6}$ ms^{-1}. Irrigation was then shut off and a 6-minute solute pulse with a concentration of 428 mg/l (Li^+) was applied at the surface before irrigation was reestablished.

The second solute experiment was conducted under unsteady-state conditions. The unsteady-state experiment was performed by applying a periodic square pulse with an average flow equal to the flow in the calibration experiment. Each pulse was 6 min long with a water flux of $1.14 \cdot 10^{-4}$ m^3s^{-1}, corresponding to a water flux of $3.92 \cdot 10^{-5}$ ms^{-1}. The pulses were applied with a 24 minute period. During the first 90 s of the first pulse, water with a lithium concentration of 405 mg l^{-l} (Li^+) were. Pure water was used during the remaining 270 s of the pulse.

Water samples were collected at the column outflow every minute up to 40 minutes after the solute was applied, every five minutes from 40 to 100 minutes, every twenty five minutes from 100 to 200 minutes and, finally, every fifty minutes. The last sample was taken at 500 minutes for the steady-state experiment and at 250 minutes for the unsteady-experiment. The samples were analyzed for concentration of Li^+ using an atomic photospectrometer. The results of the solute experiments are shown in Fig. 3.

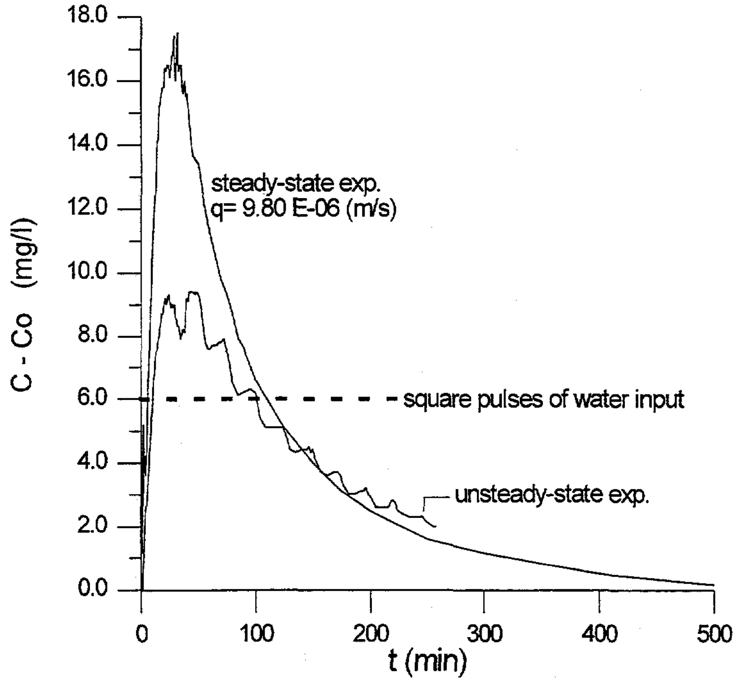

Fig. 3 Chemographs for steady- and unsteady-state solute experiments, registrated at the outflow, of the column (Z=1.2m). The time and duration of the applied water pulses are marked by bold strokes.

The observed curves show early steep rises and long tails, which support the hypothesis of fast macropore flow in a small fraction of the total porosity and a large stagnant water volume from which the solute slowly diffuses out in the mobile part of the system. The tailing is more pronounced for the unsteady-state experiment in which there is temporary storage of water due to high water flux during the pulses, which force water into voids and dead-end pores in the drainable region of the matrix . This causes the solute to diffuse deeper into the immobile region where it is not so easily recovered. The breakthrough time in the steady state experiment, defined as the time at which the breakthrough curve reaches its peak, corresponds to a volume of about 0.046 m^3 which has flowed through the system, which represents about 1.3% of the total volume, or 11% of the effective porosity. In the unsteady-state experiment, every water pulse input corresponds to a peak in the chemograph. The first two peaks are of equal magnitude and correspond to water displacement volumes of 0.043 m^3 and 0.082 m^3, respectively. If the breakthrough time is taken as the average of the first two peaks, the water displacement volume represents about 1.7% of the total volume and 14% of the effective porosity. There is apparently no significant difference between the two experiments in terms of breakthrough time. However, by fitting a log-normal distribution function to the experimental data, Rosqvist and Bendz [1998] showed that the residence time for a solute particle is twice as long in the unsteady-state experiment than in the steady-state experiment. The expected residence time determined corresponded to water displacements of 6.1% and 13.9%, expressed as a percentage of the total volume, for the steady- and unsteady-state experiments, respectively. This can probably be explained by the fact that water is forced into adjacent voids and pores during the pulses. This increases the effective area of the boundary of the macropore and facilitates diffusion of the solute deeper into the immobile domains. The active immobile domain can, accordingly, be expected to be larger in the unsteady-state experiment.

CALIBRATING THE MODEL

Steady-state experiment

The sample was at field capacity when the experiment began, so that $w_{im}^{act} + w_{im}^{pas} = FC$. Since the matrix was at field capacity and the flow was steady, no accumulation of water took place during this experiment. The S term is accordingly set to zero.

In the solute pulse domain, which is defined by the time axis, the lower boundary, $z=Z$, and the solute characteristics $z(t,0)$ (the solute front) and $z(t,T_p)$, the mobile and active immobile domains are, for the sake of simplicity, assumed to be perfectly mixed. Using the peak value of the chemograph, $\hat{C}$, the active immobile water content could thus be determined from a mass balance as:

$$w_{im}^{act} = \frac{m}{V\hat{C}} - w_m \tag{27}$$

The transfer coefficient, α, was then adjusted, and gradually increased to a point at which perfectly mixed conditions, according to the assumption above, just set in. This is the point at which the model fits the observed peak of the chemograph, so $C_{im}^{act}(t_{sf}) = C_m(z, t_{sf})$, $0 \le z \le Z$, where t_{sf} is the arrival time of the solute front. The result of the calibration procedure is shown in Fig. 4 and is summarized below:

Result of calibration procedure: $\begin{cases} \alpha = 1.5 \cdot 10^{-4} \quad s^{-1} \\ \beta = 4.0 \cdot 10^{-6} \quad s^{-1} \\ w_{im}^{act} = 0.057 \end{cases}$

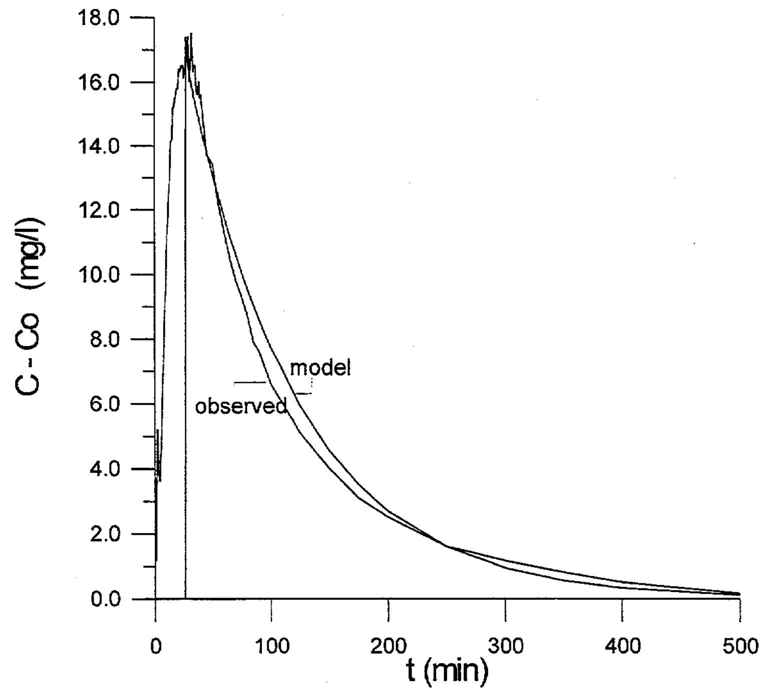

Fig. 4 Measured chemograph and calibrated model chemograph.

As can be seen in the figure, the model cannot describe the dispersion of the solute front due to the strictly convective solute flux law that was employed. The value of the diffusion coefficient governing transport into the immobile passive domain, β, was set so that the mass recovery in the model was identical to the observed mass recovery of 71%. Omitting the time period $0<t<t_{sf}$ since the model cannot handle dispersion, the coefficient of determination was calculated to be 0.77 (61 data points).

Unsteady-state experiment

Here, the S term plays the role of an exchange term and switches signs depending on the direction of the hydraulic gradient between the matrix and the channel domain. The solution of the characteristic equations (6) and (7) has been presented in Bendz et al. [1998b] for the case when a sequence of square pulses of volume flux density, q_u, with a duration T is supplied to the surface of the medium. The solution is divided into two main domains, the infiltration domain and the internal drainage domain, denoted *D1* and *D2*, respectively. A portion of the drainage domain forms a third solution domain, *D3*. The solution domains are shown in Fig.5.

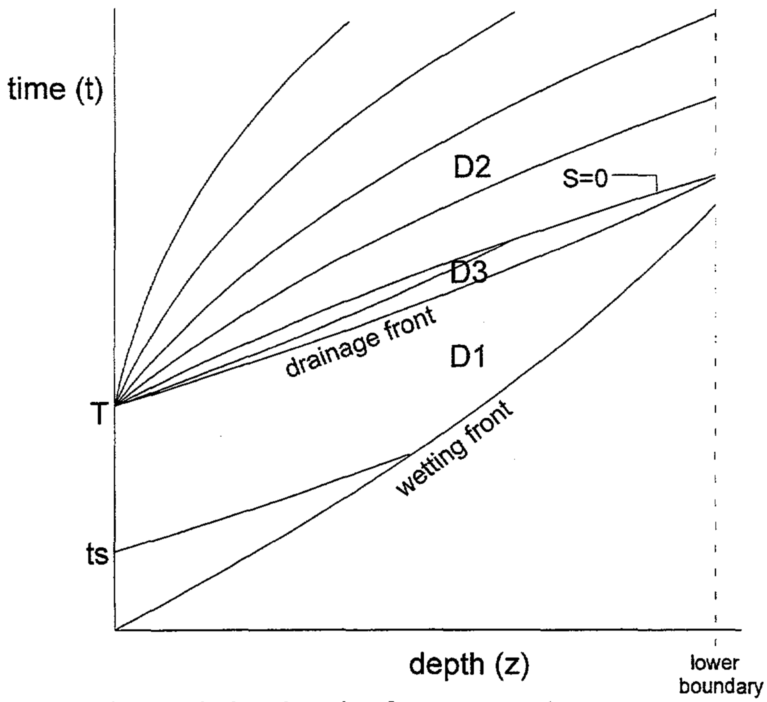

Fig.5 Solution domains for one water input event.

In the infiltration domain, *D1*, S is positive. Water is forced into voids and pores in the drainable region of the matrix due to the potential gradient that builds up during the irrigation pulses. The characteristics originate from the time axis on point t_s, where $0 \leq t_s \leq T$. In domain *D2* water is flowing back into the channel system and the S term becomes negative. Domain *D3* is a region of internal drainage, however, the water potential is higher than in the storage region so the S term is still positive. Domain $D1$ and $D3$ is divided by the drainage front that originates from the position T on the time axis and travel with the wave celerity defined by (4). A moving boundary, originating from position T on the t-axis, divides the domains $D2$ and $D3$ and carries the water content at which the mobile and immobile regions are in equilibrium, i.e., S is zero. The characteristics in $D2$ and $D3$ originate from the position T with a water content that vary from one characteristic to the other, from w_u to zero. The characteristics are therefore forming a fan shape.

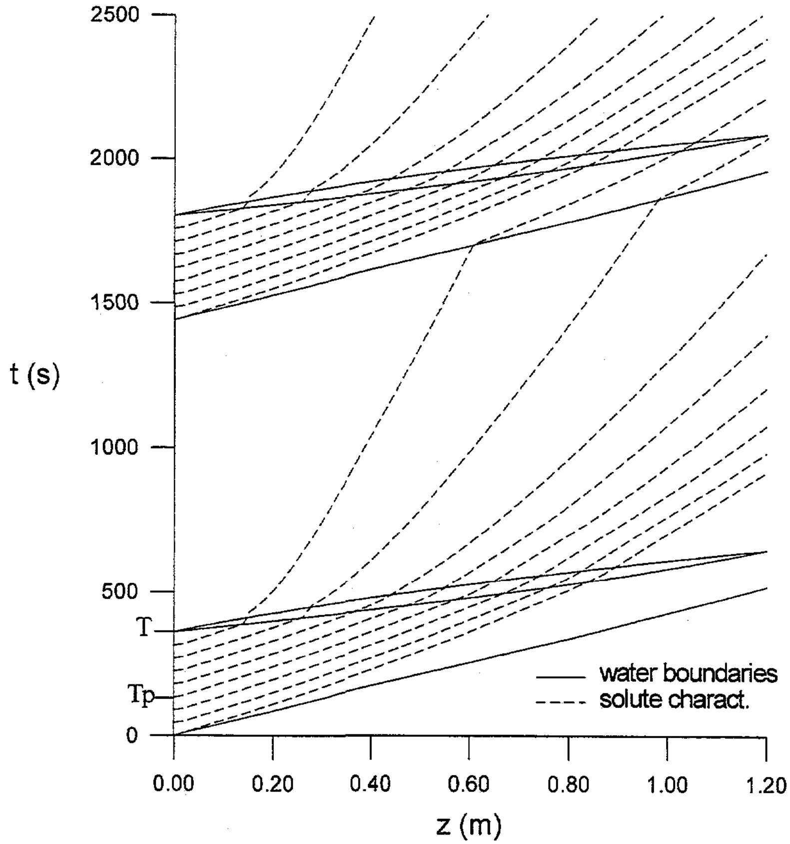

Fig.6 Solute characteristics plotted for an input of solute, duration T_p, followed by a sequence of square pulses.

Since the solute, according to equation (21), travels with the average water velocity, the characteristics of the solute transport differ from the characteristics of the water flow that were derived in our previous paper [Bendz et al. 1998b] and which are shown in Fig.2. The solute characteristics were determined by inserting an expression for the water content into equation (1) and employing a simple routing scheme using a Runga-Kutta algorithm. The wetting front, the drainage front and the moving boundary, $S=0$, for the first and second pulse are showed in Fig.6 together with the solute characteristics.

According to the previous discussion it can be expected that the fraction of the immobile domain that was active in transferring solute was significantly larger under unsteady conditions than under steady-state. This was verified by the model. By simply adjusting the immobile fraction to 0.094, a good fit to the experimental data was obtained. The coefficient of determination was found to be 0.94. In Fig. 7, the observed and modeled chemographs are plotted for 8 water pulses.

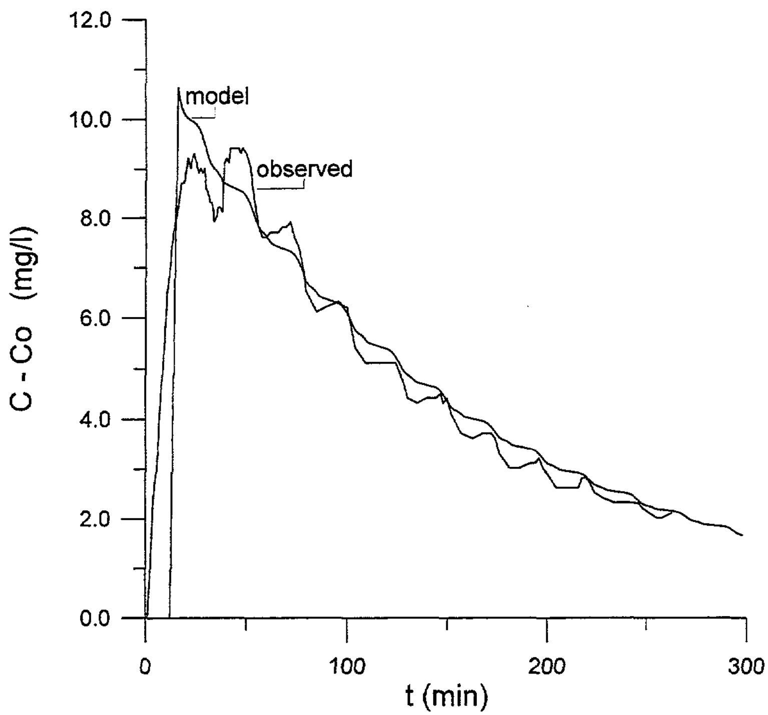

Fig. 7 Observed and modeled chemographs.

DISCUSSION

Several assumptions are built into the model, some of which have been made on an ad hoc basis and are accordingly only valid in this specific case. The most important assumptions and simplifications are listed below:

1. The ensemble of channels is lumped into a channel domain.
2. The water that is applied by sprinkling on the surface is assumed to quickly find its way to the channel domain where it rapidly flows downwards. Vertical flow in the matrix is neglected.
3. The solute used in the experiments, lithium ions Li^+, is assumed to be conservative.
4. The diffusion rate is described by a mass transfer coefficient and is assumed to be linearly proportional to the difference in concentration between the domains.
5. Spatial concentration gradients in the active and passive immobile domains are neglected, this means that each of the immobile regions is assumed to be well mixed.
6. The observed dispersion of a solute pulse applied at the surface is assumed to be solely the result of diffusional exchange of solute between the mobile domain and the immobile domains.
7. The loss of mass is assumed to be due to deeper diffusion of the solute into the passive immobile domain, where it cannot be recovered on a short-term basis.

Assumption 2 is justified due to the highly compacted and stratified nature of the waste medium and presence of impermeable surfaces (plastics), which make it likely that the applied water flows horizontally until it enter a channel where it can flow downwards. Assumptions *1*, *5*, and *6* are justified on a small scale, where spatial variability may be neglected. On landfill scale, the spatial variability must be taken into account. A good fit of the model to the observed data does not imply that the assumptions *4* and *7* are correct. Conclusions regarding transport mechanisms cannot be drawn solely on the basis of the fitting of models to experimental data [Brusseau and Rao, 1990; Jury and Roth, 1990, Sardin et al., 1991]. It can be expected that the determined diffusional mass transfer coefficients, α and β, in the presented model also include effects of mechanical mixing and sorption and desorption processes. Regarding assumptions *3* and *7*, it is important to point out that a solute can only be conservative in a theoretical sense. In practical experiments, some degree of sorption will always take place. However, due to the high degree of recovery in the steady-state experiment, sorption processes were assumed to be small in comparison with the mass transfer between the domains.

CONCLUSIONS

The three-domain conceptualization appears to be physically sound, but introduces parameters which are difficult to determine independently. All though it is difficult to use the model for predictions, it has been shown to perform well as an interpretation tool, despite the simple solute flux law. The proposed characteristic solution of the governing equation provides a didactic illustration of the solute flow paths. As long as a flow model is available which can describe the mobile water content in space and time, this solute model is easy to apply for any boundary conditions with respect to water dynamics.

It can be concluded that the residence time in the system is dominated by the size of the active immobile domain and the mass exchange with the mobile domain. By applying the model to data from solute experiments performed under both steady and unsteady conditions, it has been demonstrated that the active fraction of the immobile domain is not fixed, but is dependent on the water input pattern. The result may have a practical implication for efforts to enhance the biodegradation process in landfills by recycling of the leachate. In order secure that wetted volume becomes as large as possible it is favorable if the leachate is irrigated in pulses.

Here, only conservative solute has been considered, but the model can be extended so as to be valid for the transport of sorptive compounds by incorporating an appropriate sorption isotherm.

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