DISSERTATION

# A New Method for Modeling Water Flow and Water Storage in Municipal Solid Waste Landfills

## Eine neue Methode zur Modellierung der Wasserströmung und Wasserspeicherung in Hausmülldeponien

ausgeführt zum Zwecke der Erlangung des akademischen Grades eines Doktors der technischen Wissenschaften unter der Leitung von

o.Univ. Prof. Dr. Paul H. Brunner
Institut für Wassergüte und Abfallwirtschaft 226

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Fakultät für Bauingenieurwesen

Dipl.-Ing. Johann Fellner
Matr.Nr. 9340369
1130 Wien

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*"Nessuna umana investigazione si può dimandare vera scienza, s'essa non passa per le matematiche dimostrazioni"*

Leonardo da Vinci (1452 – 1519)

*„No human investigation can be entitled true science if it does not proceed by the way of mathematical demonstration"*

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## ABSTRACT

This thesis focuses on the numerical modeling of water movement and water storage in municipal solid waste (MSW) landfills. In particular the temporal and spatial leachate generation is considered. Hydraulic investigations at landfill sites indicate that the water flow is highly non-uniform. Preferential flow paths dominate the water transport. The non-uniform flow regime is caused by the heterogeneous character of the waste material itself, the disposal and compaction procedure, and by the construction elements such as gas wells or daily cover layers. Landfill models that incorporate preferential flow originate from soil physics and were developed for fissured or cracked soils. However, the special textural characteristics of landfills lead to different water flow patterns. Contrary to soils, water flow in landfills is funneled to favored pathways with increasing depth. Moreover, preferential flow in landfills occurs also during dry periods.

In this thesis a two-dimensional two-domain approach for modeling water flow in landfills has been developed. Thereby a flow field consisting of one vertical favored flow path (channel domain) surrounded by the waste mass (matrix domain) is defined using the software HYDRUS-2D. This model enables the calculation of water flow, solute and heat transport in porous media at variable boundary conditions. The results show that water in landfills follows a preferential path determined by high permeability and low or even no retention capacity. The bulk of the landfill (matrix domain) is characterized by low permeability and high retention capacity.

The water flow model is calibrated using data from two landfill sites in Austria (Breitenau) and Sweden (Spillepeng). Predicted leachate generation corresponds well with the observed discharge. Parameters calibrated and thus heterogeneity of the flow regime is different for the two landfills. In order to quantify the heterogeneity of the flow regime, the transport of highly soluble salts is investigated. The calibrated water flow model and HYDRUS-2D were used to simulate the solute discharge. This allows determining the fraction of waste mass engaged in water flow. For the investigated landfills this fraction varies between 25 % and 50 %.

The new model improves prediction of future emissions of MSW landfills, because it allows assessing flows and stocks of water, the key variables in landfills, in a quantitative way.

## DEUTSCHE KURZFASSUNG


Ziel der Arbeit ist die Entwicklung eines mathematischen Modells zur Beschreibung der Wasserbewegung und Wasserspeicherung in Hausmülldeponien. Das Hauptaugenmerk liegt in der Bestimmung des örtlichen und zeitlichen Sickerwasseraufkommens.

Hydraulische Untersuchungen an Hausmülldeponien zeigen, dass die Wasserverteilung innerhalb des Deponiekörpers heterogen ist. Präferenzielle Fliesswege bestimmen das Abflussgeschehen. Große Teile der Deponie sind kaum Wasser durchflossen. Verantwortlich für die ungleichmäßige Wasserverteilung sind die unterschiedlichen Materialeigenschaften und die spezielle Struktur des Deponiekörpers (lagenweiser Einbau des Abfalls, Konstruktionselemente wie Gasbrunnen und Zwischenabdeckungen). Bisherige Deponiemodelle in denen präferenzieller Abfluss berücksichtigt wird, sind an Wasserhaushaltsmodelle für Böden angelehnt. Ein Vergleich von Böden und Deponien zeigt jedoch wichtige Unterschiede. In Böden nimmt der Anteil an präferenziellem Abfluss mit zunehmender Tiefe ab, in Deponien hingegen zu. Zusätzlich erfolgt in Deponien konträr zu Böden ein Abfluss über bevorzugte Sickerwege auch während Trockenwetterperioden.

Diese besonderen Strömungsverhältnisse wurden in einem neuen zwei-dimensionalen zwei-Bereichsansatz berücksichtigt. Der Deponiekörper wird in einen feinporigen Matrixbereich mit geringer hydraulischer Durchlässigkeit und hohem Speichervermögen sowie einen vertikalen Sickerpfad mit hoher Durchlässigkeit und vernachlässigbarer Speicherkapazität unterteilt. Die mathematische Umsetzung dieses Konzeptes erfolgt mit Hilfe des Stofftransportmodells HYDRUS-2D. Dieses Programm ermöglicht es den Wasser-, Stoff- und Wärmetransport in variabel gesättigten porösen Medien unter Berücksichtigung veränderlicher Randbedingungen zu berechnen. Das Modell wird anhand von Messdaten zweier Deponien in Österreich (Breitenau) und Schweden (Spillepeng) kalibriert. Mit dem zweidimensionalen zwei-Bereichsansatz kann eine gute Übereinstimmung zwischen gemessenen und berechneten Sickerwasserabfluss erreicht werden. Unterschiede bei den kalibrierten Parameterwerten für die einzelnen Deponien können durch unterschiedliche Wasserverteilung in den Ablagerungen erklärt werden. Um ein Maß für die Homogenität der Wasserströmung zu erhalten, wird mit Hilfe von HYDRUS-2D der Austrag von leicht löslichen Salzen modelliert. Für die zwei Deponien variiert der von Wasser durchströmte Anteil zwischen 25 % und 50 %. Selbst für Deponieabschnitte, die mit ähnlichem Abfall und auf dieselbe Weise verfüllt wurden, sind erhebliche Unterschiede in der Homogenität der Wasserströmung feststellbar.

Da im Deponiekörper ablaufende Reaktionen stark vom Wassergehalt und Wasseraustausch abhängig sind, ermöglicht das entwickelte Modell anhand der Kenntnisse der Strömungsverhältnisse und des Wasserdurchsatzes den Stabilisierungsgrad und damit das zukünftige Emissionsgeschehen von Hausmülldeponien besser abschätzen zu können.

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## LIST OF ACRONYMS

| | |
|---|---|
| BV | Bed volume |
| BTC | Breakthrough time |
| CDE | Convection-dispersion-equation |
| CI | Compartment I |
| CII | Compartment II |
| CIII | Compartment III |
| COD | Chemical oxygen demand |
| FILL | Flow investigation for landfill leachate |
| HELP | Hydrologic evaluation of the landfill performance |
| L/S | Liquid to solid ratio |
| LSR | Landfill simulation reactor |
| MSW | Municipal solid waste |
| PTF | Pedo-transfer-function |
| PVC | Polyvinylchloride |
| SCS | Soil conservation service |
| SUTRA | Saturated-unsaturated flow and transport model |
| TOC | Total organic carbon |
| WS | Wet substance |
| ZAMG | Zentralanstalt für Meteorologie und Geodynamik (Central Institute of Meteorology and Geodynamics) |

## GLOSSARY

*Base flow:* The portion of the discharge that is derived from storage

*Bed volume:* The amount of pore space occupied by water

*Breakthrough curve*: The relative solute concentration in the outflow from a column of a porous medium after a step change in solute concentration has been applied to the inlet end of the column, plotted against the volume of outflow (often in number of bed volumes).

*Channel domain:* The domain representing connected fissures and preferential pathways with a pore diameter > 50 μm

*Conservative substance:* A substance whose concentration in water does not change, except by dilution (does not undergo adsorption or degradation processes)

*Field capacity:* The volumetric water content in a porous medium 2 - 3 days after being saturated (by rainfall or irrigation) and after free drainage has ceased

*Fraction of waste mass participating in water flow:* In this portion of the landfill the convective solute transport is significant (at least one magnitude) higher compared to diffusive transport (that implicates a water flux density > 0.05 mm/d)

*Gravitational Potential:* The gravitational potential of water is the amount of work required per unit quantity of water to move a very small amount of water reversibly and isothermally from a pool of pure water at atmospheric pressure at a reference level to another pool of pure water at the elevation of interest.

*Hydraulic flow regime:* The magnitude, timing, duration, distribution and frequency of water flow

*Hydraulic homogeneity:* (= uniformity of water flow) At each point within the porous medium the water flux density is constant

*Hydrodynamic dispersion*: The process wherein the solute concentration in flowing solution changes in response to the interaction of solution movement with the pore geometry of the porous media, a behavior with similarity to diffusion but only taking place when solution movement occurs

*Hydraulic homogeneity grade:* Measure for the hydraulic homogeneity expressed by the fraction of waste mass participating in water flow

*Matrix domain:* The domain representing the fine pored bulk of waste with a maximum pore diameter of 50 µm

*Matrix Potential:* The matrix potential of water in a porous medium is the amount of work required per unit quantity of water to move a very small amount of water reversibly and isothermally to the point of interest in the porous medium from a pool of pure water at atmospheric pressure at the same elevation.

*Pore water velocity:* The velocity at which water travels in pores relative to a given axis. It is equal to the water flux density divided by the volumetric water content

Preferential flow: The process whereby free water and its constituents move by preferred pathways through a porous medium

*Stabilization:* The reduction of the emission potential of the landfilled waste with the objective of final storage quality

*Total Potential:* The total potential of water in a porous medium is the amount of work required per unit quantity of water to move a very small amount of water reversibly and isothermally from a pool of pure water at atmospheric pressure and at a reference level to the point of interest in the porous medium. This is the sum of the matrix potential and the gravitational potential

*Uniformity of water distribution:* At each point within the porous medium the volumetric water content is constant

*Water flux density:* The volume of water passing through the porous medium per unit cross-sectional area (perpendicular to the flow) per unit time

## LIST OF VARIABLES

| | |
|---|---|
| $a$ | Air entry value [m] |
| $a_k$ | Dimensionless exponent [-] |
| $b$ | Campell exponent [-] |
| $b_k$ | Hydraulic conductivity (conductance) under saturation [m $s^{-1}$] |
| $+B$ | Biochemical water production [l $t^{-1}$ MSW] |
| $-B$ | Biochemical water consumption [l $t^{-1}$ MSW] |
| $c$ | Effluent solute concentration [mg $l^{-1}$] |
| $c_i$ | Inflow solute concentration [mg $l^{-1}$] |
| $c_o$ | Initial solute concentration of the pore water [mg $l^{-1}$] |
| $D$ | Dispersion coefficient [$m^2$ $s^{-1}$] |
| $ET$ | Actual evapotranspiration [mm] |
| $ET_o$ | Reference evapotranspiration (referred to grass of 12 cm height during the growing season) [mm] |
| $ET_p$ | Potential evapotranspiration (referred to considered crop) [mm] |
| $FC$ | Field capacity [$m^3$ $m^{-3}$] |
| $G$ | Vapor in gas [l $t^{-1}$ MSW] |
| $h$ | Suction (pressure) head [m] |
| $\Delta H$ | Difference in the hydraulic head [m] |
| $K$ | Hydraulic conductivity [m $s^{-1}$] |
| $K_k$ | Hydraulic conductivity [m $s^{-1}$] at the water content of $\theta_k$ |
| $K_s$ | Saturated hydraulic conductivity [m $s^{-1}$] at the water content of $\theta_s$ |
| $K^A{}_h$ | Horizontal component of the anisotropy tensor of the hydraulic conductivity [-] |
| $K_c$ | Crop coefficient [-] |
| $\overline{K}(\theta)$ | Tensor of the unsaturated hydraulic conductivity [m $s^{-1}$] |
| $L$ | Leachate amount [mm] |
| $l$ | Pore connectivity parameter [-] |
| $m$ | Cumulative discharged solute mass [g] |
| $m_o$ | Initial solute mass of the pore water [g] |
| $m_w$ | Initial mass water content (referred to wet mass) [$dm^3$ $kg^{-1}$] |
| $n$ | Porosity [$m^3$ $m^{-3}$] |
| $n_g$ | Form coefficient of the water retention function [-] |

| | |
|---|---|
| P | Precipitation [mm] |
| p | Pore coefficient [-] |
| $\nabla p$ | Pressure gradient [kg $m^{-2}$ $s^{-2}$] |
| Q | Volume flux [l $s^{-1}$] |
| q | Water flux density [m $s^{-1}$] |
| r | Radius [m] |
| R | Water added (leachate recirculation) [l $t^{-1}$ MSW] |
| S | Water storage inside the landfill [l $t^{-1}$ MSW] |
| $S_e$ | Degree of saturation [-] |
| $\Delta s$ | Length of the media through which water passes [m] |
| t | Time [s] |
| V | Surface runoff [l $t^{-1}$ MSW] |
| $V_{tot}$ | Total volume of the porous media [$m^3$] |
| v | Pore water velocity [m $s^{-1}$] |
| W | Initial mass water content of landfilled MSW [l $t^{-1}$ MSW] |
| z | Coordinate [m] |
| $\alpha$ | Form coefficient in the water retention function [-] [$m^{-1}$], inverse bubbling pressure |
| $\alpha_h$ | Pressure head scaling factor [-] |
| $\eta$ | Dynamic viscosity [kg $m^{-1}$ $s^{-1}$] |
| $\lambda$ | Empirical pore coefficient [-] |
| $\theta$ | Volumetric water content [$m^3$ $m^{-3}$] |
| $\theta_a$ | $\leq \theta_r$ extrapolated volumetric water content [$m^3$ $m^{-3}$] at infinite small matrix potential |
| $\theta_k$ | Volumetric water content [$m^3$ $m^{-3}$] at the hydraulic conductivity $K_k$ |
| $\theta_m$ | $\geq \theta_s$ extrapolated volumetric water content [$m^3$ $m^{-3}$], at full water saturation |
| $\theta_{ma}$ | Water content of macropores participating in the flow process [$m^3$ $m^{-3}$] |
| $\theta_r$ | Residual volumetric water content [$m^3$ $m^{-3}$] |
| $\theta_s$ | Saturated volumetric water content [$m^3$ $m^{-3}$] |
| $\rho$ | Density [kg $m^{-3}$] |
| $\psi$ | Total hydraulic potential [m] |
| $\psi_g$ | Gravitational potential [m] |

$\psi_m$ Matrix potential [m]

$\psi_o$ Osmotic potential [m]

$\nabla \psi$ Gradient of the hydraulic potential [m $m^{-1}$]

# 1. Introduction

Land disposal of waste has been practiced for centuries. In the past it was generally believed that leachate from waste is purified by soil and groundwater, and hence contamination of groundwater was not an issue (Bagchi, 1990). Thus, disposal of waste in the form of open dumps at all type of sites (e.g., gravel pits, ravines, etc.) was an acceptable practice until the early 20iest century. However, with increasing concern for the environment in the late 1960s landfills become under scrutiny. Within a decade several studies (Andersen & Dornbusch, 1967; Nöring et al., 1968; Zanaoni, 1972; Dunlap, 1976; Kelly, 1976) showed that landfills do significantly contaminate groundwater.

As a result of this finding steps from open dumping of wastes towards sanitary landfills were made. Regulations concerning technical equipment, site characteristics and operation of landfills were enacted and improved with time. Landfill technology has become increasingly sophisticated over the past few decades.

However, in spite of all claimed technical facilities of landfills, gradients of matter and energy between landfill and the surrounding environments still exist. 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 a long term, matter and energy will leave the landfill unless their storage is maintained by a continuous input of energy. Since long term records of the mass flow out of landfills are not available it can 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. The rate of matter leaving the landfill depends on the mass and energy gradient as wells as on the "flow resistance" between landfill and the surroundings, whereby the term "flow resistance" represents physical and chemical barriers, respectively. The aim of modern landfill management is to equilibrate the energy gradient between landfill and the surrounding environment in a controlled manner to a "final storage quality", where the emissions are considered not significantly contribute to natural substance fluxes in soils, air and water (Brunner, 1992). The landfill can become thereby an integrated part of the environment.

Existing landfills of municipal solid waste (MSW) are far from requirements of "final storage quality". Major environmental concerns associated with MSW landfills, containing high content of biodegradable organic matter, are related to the generation of leachate and biogas. Effects of leachate emissions from landfills are local for underlying groundwater and soils

(Ehrig, 1983), whereas production and emission of methane gas posses a global pollution potential since it is a greenhouse gas. It is estimated that solid waste landfills contribute 10 % of the global anthropogenic methane emissions (Watson et al., 1996). The emissions of leachate from MSW landfills will stay on an environmentally incompatible level for hundreds of years (Henseler et al. 1985; Belevi & Bacchini, 1989; Stegmann & Heyer, 1995; Krümpelbeck & Ehrig, 2000). Quantity and quality of leachate and biogas formed depend upon the characteristics of the waste, the design and operation of the landfill and the climatic conditions (temperature, precipitation, and evapotranspiration). In order to stabilize a landfill in a controlled and efficient way, so that environmental impacts are minimized from a short and long viewpoint, understanding of the processes in the landfill interior is crucial. In the last three decades water and water flow were identified as the main factors determining the metabolism of landfills (e.g. Pohland, 1975; Leckie et al., 1979; Bookter & Ham, 1982). Water is on the one hand essential for the biochemical decomposition of organic substances and on the other hand needed for leaching of soluble compounds. Different investigations (Klink & Ham, 1982; Bogner & Spokas, 1993; Christensen et al., 1996) showed that enhanced water flow through waste leads to an acceleration of biochemical processes, as water is the only carrier of substances within a landfill and only water flow facilitates the redistribution of chemicals, micro-organisms and nutrients. Water is also needed for hydrolysis which is the first steep in the anaerobic degradation process.

As water plays the key role in the metabolism, knowledge of water distribution and movement is fundamental for understanding the reactor MSW landfill. Several researchers have pointed out, that in order to improve existing models for describing the landfill behavior, further research must focus on the presence and flux of water (e.g. Straub & Lynch, 1982a; Ehrig, 1983; Augenstein & Pacey, 1991, El-Fadel et al., 1997). A better understanding of water movement inside the waste mass will benefit both the prediction of long term aftercare measurements of existing landfills and the design of strategies for accelerated stabilization, so that burdens on the future generations may be minimized (Beaven et al., 2001).

# 2. Objectives and Scope of Study

The main objective of the presented thesis is to investigate mechanisms governing water flow in MSW landfills. Based on existing approaches and conceptual considerations a mathematical model for describing transport and storage processes of water will be designed.
The development and application of this mathematical model shall enable better insights into the hydraulic behavior of MSW landfills. The calibrated flow model will help to quantify transport processes and make the flow regime in different landfills comparable.
In order to ensure that governing aspects of water movement at field scale are accounted for, the model calibration and validation is carried out using data from full size landfills.

In particular the work addresses the following questions:

- Which model concepts for describing transport and storage of water in MSW landfills have been developed so far?
- What are their benefits, drawbacks and limitations?
- What are the governing mechanisms determining water transport in MSW landfill, and how far are they included in present models?
- Is water flow in landfills comparable to those in soils (as many landfill models so far are adopted from framework carried out for soils)?
- How can leachate generation and its underlying mechanisms be described using existing mathematical formulations?
- To which extent is it possible to reproduce observed leachate generation rates from full scale landfills using a mathematical model?
- Which model parameters determine the heterogeneity of the flow field?
- How big is the fraction of waste mass participating in water flow?

The present thesis can be considered as a framework for the hydraulic analysis of MSW landfills and its description in a mathematical way.

# 3. Landfill Modeling - State of the Art

The evaluation of potential environmental pollution resulting from MSW landfills requires basic knowledge on the inter-relationship between waste materials, landfill technology, operation strategy, biochemical decompositions processes, solute transport mechanisms and precipitation processes. In other words: the governing parameters of the water and solute household of a landfill must be identified. The use of terms like "inter-relationship" and "governing parameters" is already based on an abstraction of the reality into a model. In the background of nearly all scientific considerations models are playing, albeit often unconsciously, an important role.

Models represent reproductions of chosen parts of the reality into artificial systems, so that the fundamental relations are largely held up (Atherton & Borne, 1992).

According to the form of reproduction, models are divided into:

- Physical models (models in the literal sense)
- Mathematical models

Physical models are scaled (usually smaller) and simplified reproductions of the reality, whereas mathematical models use formal descriptions (chemical, physical, empirical or statistical equations) to map the reality or the artificial system, respectively.

In the field of landfill modeling both physical models so called "Landfill Simulation Reactors" LSR (e.g. Stegmann & Heyer, 1995) and different mathematical models are used. Recently applied approaches for predicting long term processes occurring in MSW landfills make use of natural analogous (Bozkurt et al., 2001; Döberl, 2004). This method provides only qualitative results.

The capability of material models (LSR) for predicting the future emission behavior of landfills is limited, because ongoing reactions and processes can only be accelerated to a certain extent. The prevalent method to accelerate physical and biochemical processes going on in the reactors is to enhance the exchange rate of water. The yielded limited time-lapse effect however, is associated with a deliberate modification of prevailing conditions in landfills. To what extent such changes do accelerate only decomposition processes can hardly be quantified. It is generally assumed that enhanced exchange rates of water are walking along with a displacement of emission paths (Scheelhase, 1998).

The great strength of mathematical models is that slow processes ongoing over long periods can be simulated within short time. Furthermore, the ability to predict and evaluate a variety of different scenarios without the effort and expense of physical experimentation is a main advantage of these models. However, it is crucial to remember that mathematical models are idealized representations of physical processes and as such they are driven by assumptions and available input data. In order to validate and assess a model's predictive capabilities it is necessary to verify the model results through comparison with field studies.

The following section reviews in the literature reported mathematical landfill models for simulating the generation of leachate.

### *3.1. Overview of landfill models regarding leachate genesis*

Landfill models can be subdivided according to the matter modeled as follows:

- Water flow models (water balances)
- Solute transport models

Water flow models are designed to conduct water routing and determine the total amount of leachate generated. Solute transport models however, are designed to simulate leachate composition as well. Any simulation of the leachate quality requires information on the generated leachate amount. Thus, solute transport models represent an extension of water flow models.

In the past decades most mathematical models only focused on water balance considerations. The purpose of such studies was to estimate the leachate amount generated within a certain period in order to design necessary storage tanks and treatment plants at the landfill site. Recently major interest has emerged for providing better insights into the reactor landfill. Thus, many model concepts dealing with water flow have been developed, whereas solute transport approaches for landfills trying to predict leachate quality are rarely reported in the literature. This may be partly due to the complex biological, chemical and physical processes involved in landfills that make a mathematical description difficult. The quality of solute transport models, even if they are including the governing biochemical, and physical reactions, is mainly dependent on an adequate reproduction of the water flow processes.

### 3.1.1. Leachate generation models

Leachate generation models are developed and applied to predict water discharge and storage behavior of landfills as well as its migration characteristics inside the waste mass.

According to Ramke (1991) models can be divided into

- *Layer models*
- *Statistical models and*
- *Balance models*

Not included in this classification are those models that are based on the continuum approach (Bear, 1972) of a porous medium. The continuum description assumes that the boundaries between the solid, liquid and gaseous phase of a porous media can be ignored and the physical property in any phase can be described at every point. In the following models based on this concept are summarized as *continuum approach* or *potential models*.

#### 3.1.1.1. Layer models

The concept of the layer model represents the oldest mathematical reproduction of water movement in landfills. The waste body is assumed to be homogeneous and is divided into several horizontal layers. The migration of water is gradually computed from layer to layer, whereby water drainage to underlying layers occurs only when the water content exceeds field capacity FC (water amount which can be held by a porous media against gravity force). This type of water movement is known as the main wetting front.

Remson et al. (1968) were the first who introduced a water flow model for landfills based on the layer concept. The water input for the first layer is assumed to be the difference between precipitation and potential evapotranspiration, whereby the calculations where carried out on a monthly basis. If the water content of the first layer exceeds its storage capacity (field capacity FC), the excess water percolates to the layer beneath. Thus, the landfill body uniformly wets with water from the top to the bottom. Leachate is not generated until the water content in the bottom layer reaches field capacity. Water withdraw from the landfill by evapotranspiration is only considered from the top layer. An upward water movement in the other waste layers due to capillary forces is neglected. The amount of leachate generated at the landfill bottom equals the difference between precipitation and potential evapotranspiration. Due to the use of the potential evapotranspiration the model concept of Remson et al. is suitable for recultivated landfills.

Fenn et al. (1975) improved the above model concept by replacing the potential evapotranspiration through the actual evapotranspiration. Additionally surface runoff was considered. The basic principle of the water flow modeling was kept unaltered. The main advantage of both models for the user was that only few input parameters (field capacity, initial water content) that describe the waste characteristics are required.

Helmer (1974) also predicted leachate generation from landfills using the main-wetting front approach. He modified the model of Remson et al. (1968) by distinguishing between flow conditions at field capacity and below field capacity. At field capcity the water movement is calculated according to Darcy (1856). Below field capacity however, the water flow is controlled by filling the reservoir of each layer. Enhancements of this first concept (Helmer, 1977) include a so called "base leaching" which represents a water discharge even if the water content is below field capacity. The rate of discharge depends on the actual water content of the considered layer. By means of "base leaching" heterogeneities inside the landfill as well as local water saturation should be regarded. The computations were carried out on a daily basis.

At the same time like Helmer Franzius (1977) developed a complex layer model to simulate the water flow in landfills. The starting points for his investigations were experiments in the laboratory using small cells filled with solid waste. He determined the influence of the emplacement density on the hydraulic and hydrologic characteristics of the waste material (field capacity, hydraulic conductivity, infiltration rate and actual evapotranspiration). The attained findings of these experiments concerning relations of different parameters were incorporated in his model approach. Franzius assumed that leachate is not generated until the whole landfill body reaches field capacity. The water flow at field capacity is calculated according to Darcy (1856) using the hydraulic conductivities determined in laboratory experiments. The introduced model is applicable for landfills that are under operation and landfills after closure. Franzius made the following assumptions for his model concept:

- homogeneous landfill body
- uniform water distribution
- constant water storage capacity

Although these assumptions are incorrect for landfills Franzius (1977) never discussed this issue.

The most common model for estimating leachate generation from landfills HELP is also based on the layer concept. The first version of the software code HELP (**H**ydrologic **E**valuation of the **L**andfill **P**erformance) was introduced by Schroeder et al. (1984a, b) at the U.S. Army Engineering Waterways Experiment Station. Up till now several further versions of the original code have been evolved. However, changes of the model concern the implementation of various capping systems or the adjustment to certain climatic conditions only, the basic flow equations stayed unaltered (Schroeder et al., 1994; Berger, 1998).

The original code enables the calculation of:

- surface runoff from the landfill cover according to the Soil Conservation Service (SCS)-curve number method (1973)
- actual evapotranspiration after the modified Penman equation (Ritchie, 1972),
- vertical water movement under saturated and unsaturated conditions and
- leachate discharge at the base sealing of the landfill

The vertical water flow is calculated using a modified form of the Darcy equation (1856), assuming that the hydraulic conductivity is directly proportional to the water content in the single layers. In case that the water content drops below the field capacity the hydraulic conductivity becomes zero. The input parameters required for the model are: precipitation, temperature, solar radiation, humidity at the landfill site, as well as saturated hydraulic conductivity, porosity, field capacity, wilting point of the landfilled waste and the cover layers, respectively. Nowadays after recognizing the complex hydraulic system landfill body and its insufficient reproduction by layer models, HELP is mainly used to estimate the water balance of landfill covers (Ramke, 1991; Berger, 1998).

The main advantages of layer models are the limited number of parameters describing the waste material as well as their easy determinability, and the simple mathematical formulation together with the attempt to incorporate the layer structure of the landfill into the model (Hartmann, 2000). The application of these models enables the operator to estimate the water balance parameters of landfills. However, the capability of layer models for predicting the temporal discharge of leachate must be questioned due to the insufficient assumption of a homogeneous landfill with uniform hydraulic characteristics (conductivity and storativity). Varying storage capacity as well as preferential flow paths are ignored which results (compared to observation at landfills) in overestimating the time for water to discharge from the landfill. Bengtsson et al. (1994) determined by means of the layer model concept that a duration of 10 years is required for the bottom layer of a landfill of 10 m height to reach field capacity and therefore generate leachate. Observations at landfill sites indicate shorter periods. Leachate is already generated short time after waste is landfilled.

#### 3.1.1.2. Statistical and empirical models

Statistical models build or make use of relations between input and output parameters of the investigated system, thereby neglecting internal processes.

One of the first statistical approaches in the field of landfill modeling was introduced by Ehrig (1978). Observed data of weekly precipitation, leachate discharge and potential evapotranspiration of four different landfills over a period of one year were used. Ehrig performed regression analysis using these three parameters. He found a close match between predicted and observed leachate discharge assuming a linear dependency between the actual leachate discharge and the precipitation in the week before. The potential evapotranspiration had to be neglected. Instead of this parameter a seasonal variable was introduced to represent evapotranspiration and storage processes inside the landfill body.

Based on the study of Ehrig (1978) Ossig & Tybus (1986) analyzed precipitation, evapotranspiration and leachate data from eleven landfill sites. The temporal resolution of the observed data varied from days to weeks. The regression analyses based on a daily basis resulted in less agreement compared to analyses carried out on a weekly basis. Ossig & Tybus did not discuss the reason for this result. However, it was shown that water retention characters of different landfills varied significantly (represented by diverse relations between precipitation and leachate discharge).

Jourdan (1981) applied the unit-hydrograph method (Dooge, 1959), which is commonly used in the field of hydrology, to ascertain a correlation between precipitation and leachate generation. Based on single rainfall events and thereby induced discharges the so called unit-hydrograph was determined. A unit-hydrograph gives for a considered hydraulic system the relationship between a single rainfall event of certain duration and the induced discharge. Jourdan neglected in his investigations the base flow of leachate (unaffected by precipitation input), as his aim was to describe extreme precipitation events in order to design drainage and storage facilities for the generated leachate. The application of the determined unit-hydrograph is limited to the investigated landfill and the conditions present during its analysis (cover layers, waste amount, ...). Another drawback beside the limited application is the fact, that the current situation regarding the water storage does not have any influence on the results.

A main disadvantage of statistical models is that they are limited to the specific experiment for which they have been developed. They cannot be extrapolated to simulate other field conditions. Additionally no information about impacts of the landfill body or the operation strategy on the discharge characteristics can be gained. Using this modeling concept it is impossible to get a better insight into leachate formation mechanisms.

#### 3.1.1.3. Balance models

Balance models consider input, output and storage of water in the hydrologic system landfill.

First water balances for landfilled waste (e.g. Quasim & Burchinal, 1970; Fungaroli & Steiner, 1971) were carried out at laboratory columns, where boundary conditions are exactly definable. Water was added by irrigation and evaporation from the waste was prevented by covering the columns. The amount of water stored inside the waste was either determined by weighing the whole column or by sampling and analyzing the waste at the end of the experiment.

Spillman and Collins (1986) reported water balance calculations for landfilled MSW, that incorporate evaporation respectively evapotranspiration. Their concept was based on a linkage between generated leachate and climatic water balance. Data base for their investigations represent measurements carried out at waste lysimeters over a period of five years. Leachate discharge was calculated performing the difference between precipitation and actual evapotranspiration, whereby the actual evapotranspiration was obtained using the potential evapotranspiration and an estimated maximum water storage capacity within the upper waste layers. The water balances were carried out on a weekly basis.

Baccini et al. (1987) conducted element and water balances of municipal solid waste landfills. Unlike conventional investigations considering precipitation, evapotranspiration and leachate only, Baccini et al. accounted for water storage and water input caused by the water content of the landfilled waste. Other parameters like water production and water consumption due to biological degradation processes were identified to be negligibly small. The amount of water stored in the landfill was determined measuring the water content of the landill at undisturbed drilling cores. These measurements indicate that the water storage inside the landfill stays constant over longer periods (years) and equals the initial water content of the landfilled waste material. Baccini et al. applied their balance concept to four landfill compartments of different age. In addition to the water balance considerations material balances were made up for 12 elements (C, N, P, S, Cl, F, Fe, Zn, Pb, Cd, Hg, Cu) resulting in so called transfer coefficients for each element. The transfer coefficient gives the partitioning of the element flux into the gas and liquid phases, respectively.

Water balance models are appropriate to predict the cumulative amount of leachate generated over longer periods. In case of known leachate discharge the concept enables to estimate the amount of water stored inside the waste mass. However, impacts of single rainfall events on the leachate generation cannot be evaluated. Analogous to statistical models hydraulic characteristics of the landfilled waste are not included in this model concept. Nevertheless balance models represent sophisticated tools to identify the governing in- and output parameters of investigated systems.

#### 3.1.1.4. Continuum approach models (potential models)

The continuum approach or potential models are based on the theory of saturated and unsaturated water flow in porous media. They originate from the field of soil science. Assuming that water flow in landfills resembles the water movement in soils, the potential concept was increasingly applied for water flow simulations in landfills. The concept postulates that every movement of water is caused by differences in the hydraulic potential, whereby the total potential $\psi$ of water present at a certain location consists of the matrix potential $\psi_m$ (caused by adsorption and capillary forces) and gravity potential $\psi_g$. Under fully saturated conditions water flow is computed according to Darcy (1856), whereas for unsaturated conditions Richards equation (1931) is applied. Both equations require information about the hydraulic characteristics of the considered porous media (e.g. saturated/unsaturated hydraulic conductivity, water retention characteristics). A general description of the flow equations is given in section 3.2.2.

Reported continuum approach models for describing water tranport in landfills are often combined with solute transport models, in order to predict both, leachate quantity and quality. "Potential" models dealing exclusively with water flow represent exceptions.
The first generation of water and solute transport models considered the landfill body as a homogeneous isotropic media. The reported models differ in the incorporated boundary conditions, in the considered decomposition processes and the mathematical reproduction as well as in the applied solution algorithm for the governing equations. Since the main flow formula is a partial differential equation of $2^{nd}$ order for which an analytical solution exists only for certain boundary conditions, numerical methods (Method of Finite elements or finite differences) are usually applied to solve this equation.

The first landfill model based on the continuum approach was introduced by Straub & Lynch (1982a, 1982b). The model enabled the operator to simulate water flow and solute transport, dissolving and decay of contaminants in unsaturated sanitary landfills. Water flow through the waste mass was calculated according to Richards equation (1931). The migration of dissolved contaminants is due to convective, dispersive and diffusive transport phenomena. Basic equation for computing these transport processes is the convection-dispersion-equation CDE (Lapidus & Amundson, 1952). Decomposition and dissolving processes of solid waste matters were simulated using first order kinetics. The developed model was applied to simulate data from lysimeter studies. The reported results indicated a fairly good agreement between predicted and observed data, when adjusting the input parameters.

Korfiatis et al. (1984) predicted water flow through refuse of laboratory columns using the theory of unsaturated flow through porous media. The calculations were carried out using Richards equation (1931). Based on the results of their investigations they concluded that little contribution could be attributed to capillary diffusivities when the water content exceeds field capacity. Conversely, diffusion will dominate at water contents below field capacity. Within the scope of model verification the hydraulic characteristics of the waste mass were determined in laboratory tests. The introduced model enabled to simulate the general dynamics of leachate discharge from small scale waste columns.

Based on the investigations of Straub & Lynch (1982a, 1982b) and Korfiatis et al. (1984), Demetracopoulos et al. (1986) developed a mathematical model which incorporates water and solute transport processes in MSW landfills. Compared to the previous studies the numerical techniques for simulating leachate generation and contaminant transport were improved. Another modification of the model concerned the mathematical description of the biochemical decomposition processes. Monod kinetics (1949) was used to characterize these processes. The model enabled to reproduce of the general shape for concentration history in lysimeter studies. The concentration values were shown to be sensitive to microbiological kinetic parameters. Model calibration was difficult due to the limited data on kinetic parameters and virtually none on mass transfer from the solid to the liquid phase. No comparison with actual field data was presented.

Two similar models that operate with the same basic equations were introduced by Lee et al. (1991) and Lu & Bai (1991a, 1991b). Additionally (to the model of Demetracopoulos et al., 1986) boundary conditions for the water flow module like surface runoff and evapotranspiration were taken into consideration. Leachate quality was simply expressed as concentrations of chemical oxygen demand COD. The models were used to simulate data from lysimeter experiments. One difficulty of the model application was the need for many input parameters that are usually not readily available. Indeed a sensitivity analysis showed that at least eight parameters strongly effected the model simulations. Parameters, for which the best simulation results were reportedly obtained, were physically unrealistic.

Vincent et al. (1991) presented a model to describe leachate generation, contaminant transport and biodegradation in landfills. Analogous to the above models, the leachate migration was simulated as an unsaturated flow in a homogeneous medium applying Richards equation (1931). First order kinetics was used to represent dissolving and decay of organic materials. Anaerobic degradation was reproduced by the Herbert kinetics model (a modified form of the Monod model). Leachate quality was evaluated in terms of total organic carbon content TOC only. The model was used to simulate experimental data obtained from waste columns.

Noble & Arnold (1991) developed a one-dimensional finite difference model (FULLFILL) to evaluate water transport and distribution in landfills. The theory of unsaturated flow in porous media built the basis for their model concept. Although the model incorporates biodegradation kinetics to assess leachate characteristics, the emphasis is on water movement within the waste layers. Experiments were conducted in conjunction with the modeling effort to obtain calibration data. The simulation results indicated a fairly good fit with observed data. However, the application was limited to small scale experiments.

Ahmed (1992) presented the two-dimensional unsteady state flow model FILL (Flow Investigation for Landfill Leachate) to simulate the leachate transport in landfills. The model solves the unsaturated-saturated flow equations in a two-dimensional flow field, incorporating surface runoff, and applies the Philip's equation (1969) to compute infiltration rates. The model was applied to simulate the leachate generation in a section of a landfill. Although a good fit with field data was reported, the application of the model is limited to quantify the total amount of leachate generated. Khanbilvardi et al. (1995) compared results obtained by FILL with the landfill model HELP (Schroeder et al., 1984a, 1984b). FILL reportedly indicated a lower value of leachate outflow compared to values obtained by HELP. Although FILL may better represent the field conditions, it is not clear which model provides better estimates because of the uncertainties in its parameters. Leachate quality was not addressed in FILL.

Al-Soufi (1992) introduced a three-dimensional model for simulating water flow and solute transport in soils and applied his concept to landfills. The model addressed saturated and unsaturated water flow, evapotranspiration, plant interception and overland flow from the landfill surface. Chemical and biological processes were represented using simplified empirical equations. The model was used to simulate data from experimental studies.

Although the introduced concept provides a comprehensive framework to predict leachate behavior in landfills, it suffers from the assumption of a homogeneous water distribution and the need of many input parameters.

Al-Yousfi (1992) developed the PITTLEACH model to predict the leachate quantity and quality, together with biogas generation at sanitary landfills (cited in Al-Yousfi & Poland, 1998). Analogous to other models leachate migration was described as unsaturated flow in a homogeneous porous media applying Richards equation. The infiltrated amount of water at the landfill surface was estimated performing a hydrological balance. Anaerobic decomposition of organic waste matter followed three sequential biochemical reactions (hydrolysis, acidogenesis, and methanogenesis). The model was used to simulate several experimental studies. Although it is one of the few modeling efforts that combined gas and leachate generation in one model, leachate and gas transport were modeled independently rather than as a coupled two phase problem.

With the main objective of modeling the landfill gas production Lee et al. (1993) developed the so called LEAGA-I model. LEAGA-I is a one-dimensional model that incorporates a flow module (for unsaturated vertical flow) and a biochemical reaction module, which permits the prediction of methane and carbon dioxide production. Analogous to Al-Soufi (1992) the decomposition processes are subdivided into three sequential reactions (hydrolysis, acidogenesis, and methanogenesis). The model was applied to waste lysimeter tests only.

A concept for a complex landfill model which tries to account for interactions between water movement and gas transport was presented by Swabrick et al. (1995). The intended multiphase flow model should permit the simulation of water, gas and heat transport in sanitary landfills. The model concept is based on the theory of multiphase flow in porous media (Nguyen, 1982). It operates with mass balances for each phase, whereby the single phases are coupled by simple source and sink terms. Water and gas flow through the waste mass were supposed to be calculated according to Darcy (1856). The transport of substances within these two phases should be simulated differently. In the water phase only convective migration is regarded, whereas in the gas phase convective and diffusive transport phenomena are considered. The heat transport should be calculated considering convection and diffusion of heat within the gas and water phase. Additionally to the flow module Swabrick et al. (1995) planned to incorporate a biochemical reaction module that should enable to simulate the anaerobic decomposition of the organic matter. Contrary to other developed models the degradation processes should be included by applying chemical balance reactions. However, the reported concept has not been realized so far into an applicable form.

A complex landfill model including various processes has recently been developed at the Technical University of Braunschweig (Haarstrick et al., 2001; Hanel, 2001; Kindlein et al., 2003). The model incorporates water flow, solute transport, gas movement and heat transport in three dimensions. The main focus however, is on biochemical digestion of the organic matter. In order to study the impact of environmental factors on the biological decomposition processes, pH, temperature and hydrogen changes have been integrated into the degradation model as reaction influencing terms. Water flow in the landfill is calculated using a generalized Darcy law for multiphase flow (Bear, 1972). The waste mass is considered to be homogeneous and isotropic. The movement of substances within the liquid and gaseous phase is attributed to convective and diffusive transport mechanisms. Altogether the introduced model requires 22 input parameters, whereby all of them have been taken from the literature. No calibration or validation using observation data from landfills has been performed yet. Hanel et al. (2001) however, compared model results with data obtained from small scale landfill simulation reactors. They concluded that the general trend of the observed processes could be reproduced quite well applying the mathematical landfill model. In order to reduce the required amount of input parameters, Hosser et al. (2003) developed a method based on the reliability theory which enables to filter the important parameters. Nevertheless the model has not been calibrated and validated at a field scale so far.

All models mentioned above that are based on the continuum approach assume the landfill body to be a homogeneous media, resulting in a uniform water distribution. The assumption of a uniform flow regime is probably valid for laboratory experiments, but not for full scale landfills as tracer experiments show. This postulation is supported by the fact that a good match between observed and predicted (assuming a homogeneous flow field) leachate discharge was only achievable for small scale experiments. However, as different field investigations showed, water distribution in landfills is far from being uniform (Wiemer, 1982; Blight et al., 1992). The water content varies from saturated conditions to complete dryness. This is explained by preferential flow paths that short a large bulk of the landfill. The importance of this phenomenon is undisputed (e.g. Ehrig, 1983; Stegmann & Ehrig, 1989, Zeiss & Major, 1993), but it has been disregarded in most landfill models. Another process that is hardly incorporated in models assuming a homogeneous flow regime is the middle-term storage of water within the landfills. In particular the combination of rapid water transport (immediately after rainfall events) and storage of infiltrated water is impossible to achieve. Indeed, observations at several landfill sites (e.g. Döberl et al., 2002; Åkesson & Nilsson, 1997) prove the importance of these two transport phenomena.

To overcome the limitations mentioned above the landfill body was not considered as a homogeneous media. Attempts were made to divide the landfill into domains with different hydraulic characteristics. The terms of two- or multiply-domain models are used in this context. The consideration of "preferential" flow within the landfill was facilitated.

Within the scope of landfill modeling Young & Davies (1992) were the first who suggested the concept of a two-domain water flow. The flow field (the landfill) was supposed to be divided into a macro- and a micropore-domain with different hydraulic properties. Water flow is calculated separately for each domain applying Richards equation (1931). The main focus of the intended model was given to the degradation processes of organic matter. In particular the generation of landfill gas and its governing factors were described. The model however, did not progress from its initial stage of development. No comparison of the concept with field data is available.

Zeiss & Major (1993) investigated the water flow pattern in small waste columns which were equipped with flow sensors. Even at high water infiltration rates only 28 % of the sensors installed displayed movement of water. The results of their investigations clearly indicate that the water flow is strongly affected by channeling and preferential flow paths, even in small scale experiments. Therefore Zeiss & Major recommended that the waste default values should be revised in developed models, while new leachate generation models need to account for channeled flow (Zeiss & Uguccioni, 1994).
Based on these findings Uguccioni & Zeiss (1997) compared two different approaches to simulate water transport through MSW. They applied the one-dimensional layer model HELP and the two-domain flow model PREFLO (Workman & Skaggs, 1990) for fractured porous media to predict the leachate generation from pilot scale test cells with an average waste volume of 4 $m^3$. PREFLO assumes that the rapid water flow in the channel domain follows Poiseuille's Law (1841) and the lateral water transfer from the channels into the matrix occurs according to Richards' Law (1931). Though PREFLO seems to display the flow processes

physically more realistic than HELP, both models were unable to predict the exact shape of the observed hydrographs. Dependent on the chosen parameter values either the simulated breakthrough time (initial leachate generation) was too long or the cumulative water discharge too high. Due to these unsatisfactory simulation results, Uguccioni & Zeiss (1997) called for a new two-domain model approach that reflects channel and matrix flow.

Bendz (1998) investigated the leachate discharge from landfills and recognized the importance of heterogeneities (particularly fissures and channels) for the water movement. In order to take these heterogeneities into account he introduced a two-domain flow concept composed of channel and matrix domain. Contrary to the approach of Young & Davis (1992) the water flow in the channel domain (macropores) is computed applying the kinematics wave equation according to Beven & Germann (1981). For the matrix domain (represents micropores) Richards equation is used to describe the water movement. The interaction between both flow domains is regarded by simple source and sink terms. In addition to water transport Bendz & Singh (1999) incorporated solute transport processes into the model, whereby only convective movement ("piston flow") is considered. Between the two domains diffusive transport of solutes can take place. Only conservative substances were considered. Tracer experiments in the laboratory were conducted in conjunction with the modeling effort to obtain data sets for the calibration of the model. The concept enabled to simulate the leachate generation and the tracer breakthrough in small scale experiments. An upscaling of the introduced model to real landfill size has not been performed.

Obermann (1999) introduced a one-dimensional mathematical flow model (WATFLOW) to simulate water flow through landfills containing pre-treated waste. Special interest was given to the impacts of the following factors on the water household: landfill geometry, waste pre-treatment and landfill operation. The flow field is again divided into two domains (micro- and macropores). Analogous to Young & Davis (1992), flow in both domains is calculated applying Richards equation. The interaction between the flow domains is different however. According to Obermann water transport within the macropores occurs only if all micropores are fully saturated (threshold concept). The model accounts for varying climatic conditions, increasing landfill height, load dependent settlements and density dependent hydraulic characteristics. Obermann took the variability of the input parameters into account by performing Monte Carlo Simulations (several computations with varying parameters, the result is a possible range of values). WATFLO is limited to water transport. An enhancement of the model concept was presented by Danhamer (2002) who incorporated degradation processes of the organic matter and additionally gas and heat transport.

McCreanor & Reinhart (1996, 2000) applied the model SUTRA (Saturated-Unsaturated flow and **TRA**nsport model) to simulate the water routing in a leachate recirculating landfill. SUTRA (Voss, 1984) was developed to compute water flow and solute transport in two-dimensional variably saturated porous media. The model is based on the Richards equation and the convection-dispersion-equation.
Modeling efforts were done assuming homogeneous anisotropic and heterogeneous waste masses. The heterogeneous waste mass was simulated by applying statistical relationships to the distribution of hydraulic conductivity assigned to regions of the waste mass. The model estimations were verified using data obtained from full scale leachate recirculating landfills. Results from the verification effort indicated that channel flow is the major leachate transport mechanism. However, in order to simulate the leachate generation of MSW landfills in an appropriate way slow Darcian flow need to be considered as well.

Hartmann (2000) evaluated in his thesis the capability of different water flow model concepts (one- and two-domain concepts) to simulate the discharge from bottom ash landfills. Thereto two existing flow models were investigated: HYDRUS-1D (Šimunek, et al., 1998), a conventional potential model (one-domain model) and MACRO (Jarvis, 1991), a two-domain model. Using the two-domain model MACRO Hartmann reported a good fit with field data observed at a bottom ash landfill in Switzerland. Applying HYDRUS-1D however, a good match was only achieved using unrealistic input parameters. The concept of MACRO is based on a partition into macro and micropore-domain. Both domains operate as separate, though interacting, flow regions, each characterized by a degree of saturation, a conductivity and a flux. Richards equation is used to calculate water fluxes in the micropores, whereas the kinematics wave equation according to Beven & Germann (1981) is applied to determine the rapid water flux in the macropores. HYDRUS-1D uses only Richards equation to predict water movement. Contrary to MACRO preferential flow cannot be considered with this model. Hartmann determined the hydraulic characteristics of the bottom ash applying "pedo transfer functions" (PTF) and using information on the grain size distribution. He successfully calibrated and validated MACRO using an annual series of leachate discharge from a bottom ash landfill. However, a close similarity between the water flow pattern in fine grained bottom ash and untreated heterogeneous MSW must be questioned.

The concept of a dual-permeability model seems to be more realistic, though the multitude of required parameters complicates the calibration procedure.

Summarizing the review of reported mathematical landfill models it can be stated that, although several models were reportedly successful to a limited extent in simulating simple cases under well defined and controlled conditions, predicting leachate generation from MSW landfills can still be considered in a developmental stage. The majority of the introduced models neglects the heterogeneous nature of landfills. In rare occasions when heterogeneous flow conditions are considered (two-domain approach) model application was restricted to laboratory tests only. A framework for scaling up the models so that they remain valid on field scale, where the spatial variability must be taken into account, has not been proposed. Furthermore, reported two-domain concepts are invariably derived from the framework that has been carried out for non-uniform water flow in soils. Compared to soils landfills are more heterogeneous, which results in a bigger fraction of preferential flow. This fact was taken into account by adjusting decisive parameters. The underlying assumption however, the similarity between water flow in landfills and soils has not been justified or even discussed yet.

## 3.2. *Mathematical description of water flow in porous media*

### 3.2.1. Hydraulic analogues of municipal solid waste landfills

Landfilled MSW is a porous medium with solid material and porespace distributed throughout the volume. The pore space is filled by water and/or gas, whereby the gas can be composed of generated landfill gas and/or intruded air. In general a landfill can be treated as a three phase system, consisting of a solid phase, a liquid phase and a gaseous phase (Figure 3-1).

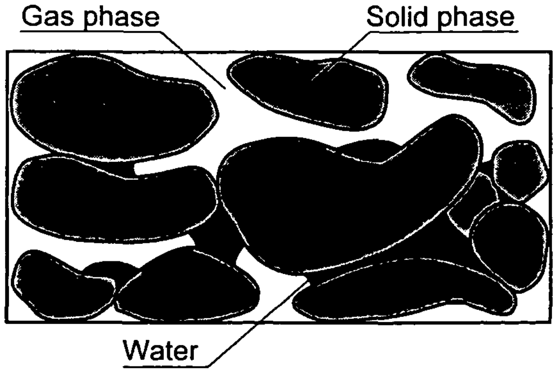



***Figure 3-1*** ***Three phase system (e.g. landfill, soil)***

The porous media which is most comparable to solid waste landfills concerning its structure, porosity, water and gas content, is soil. This finding has already been accounted for in present water flow models for landfills (see chapter 3.1.1), as all introduced concepts so far are derived from the framework which has been carried out for soils. However, in spite of general similarities between both porous media, the composition of landfills is more heterogeneous (e.g. grain size distribution, shape and hydraulic properties of single grains, organic matter) compared to soils. This heterogeneous character of landfills leads to a highly non-uniform distribution of water. Saturated zones and completely dry zones are found next to each other. The water flow in landfills is dominated by preferential pathways that short a large bulk of waste (e.g. Zeiss & Major, 1993).
Recently developed mathematical landfill models, that incorporate preferential flow (two-domain approaches, see chapter 3.1.1.4), try to account for the heightened heterogeneous characteristics of MSW landfills by simply adjusting decisive parameters.

Beside the pedosphere also the lithosphere represents to some extent a comparable medium for describing the hydraulic behavior of sanitary landfills. In particular the phenomenon of preferential flow is partly explained using analogues in the lithosphere. Döberl (2004) compares the hydraulic system landfill with karst formations that show similar discharge characteristics. In karst systems so called karst tubes (representing preferential flow paths) are accountable for the quick response of discharge after precipitation. Whereas the fine-grained dolomite releases water slowly and therefore provides discharge during dry periods ("base-flow").
Figure 3-2 gives an example for the hydrology of karst systems. Despite comparable discharge characteristics of karst formations and MSW landfills, genesis and development of preferential flow in both media is different. In karst systems favored flow paths are formed by dissolution of soluble rocks. In landfills the heterogeneity of the waste disposal is responsible for flow channels.

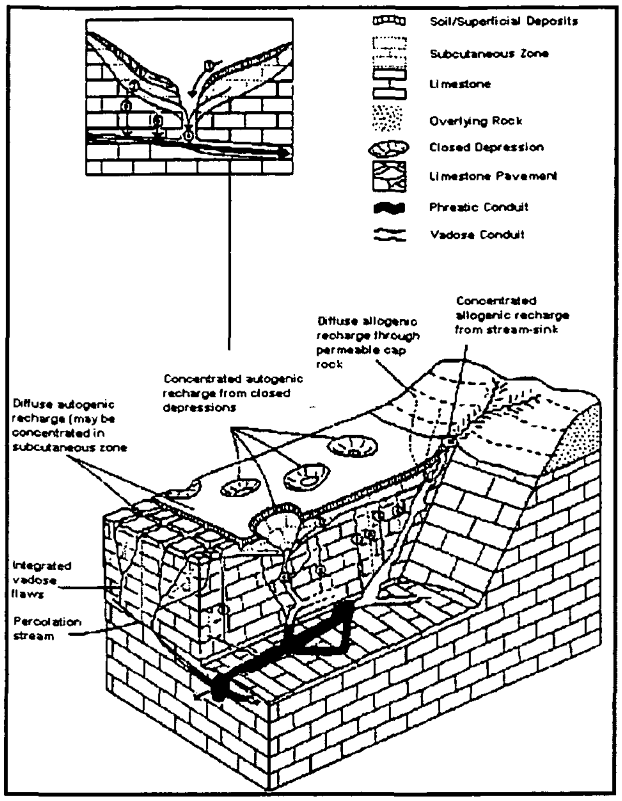



*Figure 3-2 Hydrology of karst system (Crawford Hydrology Laboratory, www.dyetracing.com)*

Attempts to model the karst hydrology range from simple Darcy flow assumptions with an average rock permeability, over two-domain approaches (rock permeability and conduit permeability) to so called discrete models, that require explicit information about the spatial extent of each conduit (White, 2002). The discrete modeling concept is inapplicable for landfills, as an exploration of single flow paths in landfills is impossible. The other two model approaches (Darcian flow, two-domain flow) correspond to the concepts for simulating water flow in homogeneous respectively fissured soils. These approaches have already been applied for sanitary landfills (see chapter 3.1.1.4).

### 3.2.2. Equations describing the water flow in porous media

In the following a short overview of mathematical equations for describing water flow in porous media is given.

Darcy (1856) made an important discovery for ground water flow. He proved in his experiments a linear relationship between the water flux density through a saturated sand column and the forces acting on the water.

*Darcy equation (one-dimensional flow):*

$$q = -K_s \frac{\Delta H}{\Delta s}$$ ***Equation 3-1***

| | |
|---|---|
| $q$ | *Water flux density [m s⁻¹]* |
| $K_s$ | *Saturated hydraulic conductivity [m s⁻¹]* |
| $\Delta H$ | *Difference in the hydraulic head [m]* |
| $\Delta s$ | *Length of the media through which flow passes [m]* |

When the water flow is three-dimensional the equation can be generalized to:

$$q = -\overline{K} \cdot \nabla \psi$$ ***Equation 3-2***

| | |
|---|---|
| $\overline{K}$ | *Tensor of the saturated hydraulic conductivity [m $s^{-1}$]* |
| $\nabla \psi$ | *Gradient of the hydraulic potential [m $m^{-1}$]* |

The Darcy equation is applicable for water flow under the condition that all pores are filled with water. Combining Darcy equation and the approach of Buckingham (1907), which assumes that the hydraulic conductivity is function of the water content, with the law of conservation of mass results in *Richards equation* (1931). This equation is applied to calculate the water flow in variably saturated porous media.

$$\frac{\partial \theta}{\partial t} = \nabla\left(\overline{K}(\theta) \cdot \nabla \psi\right)$$ ***Equation 3-3***

| | |
|---|---|
| $\theta$ | *Volumetric water content []* |
| $t$ | *Time [s]* |
| $\overline{K}(\theta)$ | *Tensor of the unsaturated hydraulic conductivity [m$s^{-1}$]* |

Richards equation assumes that the unsaturated hydraulic conductivity K(θ) is a function of the water content θ, which is again a function of the matrix potential $\psi_m$. The matrix potential

$\psi_m$ can be understood as a measure for the intensity of water fixation inside a porous media. It represents the capillarity of the single pores. The total hydraulic potential $\psi$ of water inside a porous media is defined as the sum of matrix potential $\psi_m$ and gravitational potential $\psi_g$. In rare occasions also the osmotic potential $\psi_o$ of the water can be of importance (e.g. root water uptake).

$$\psi = \psi_m + \psi_g \qquad \textit{Equation 3-4}$$

The matrix potential ranges from 0 cm, which represents water saturation of the pore space, up to $-10^7$ cm. High absolute values of $\psi_m$ correspond to low water contents. The relationship between water content and matrix potential is known as water retention characteristics. Figure 3-3 gives an example for the retention characteristic curve of a soil.

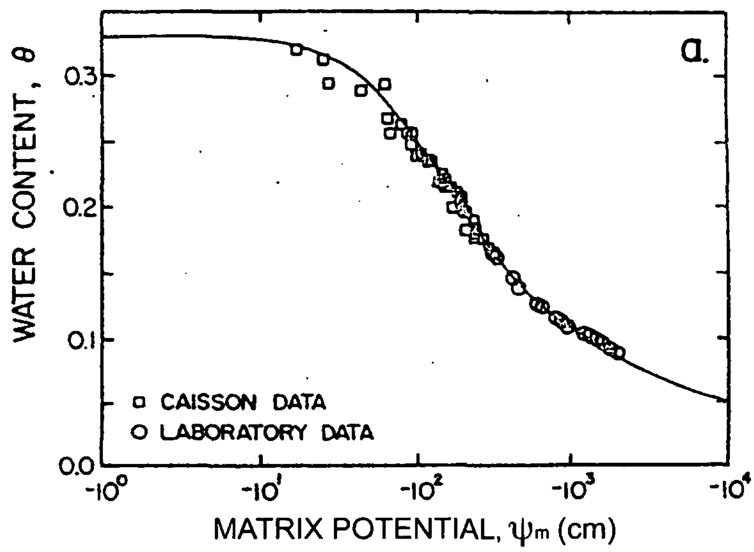



***Figure 3-3*** ***Water retention curve of a soil (van Genuchten & Šimunek, 2002)***

Several empirical equations describing the retention characteristics of soils have been developed (e.g. Brooks & Corey, 1964; Campell, 1974; Hutson & Cass; 1987; van Genuchten, 1980). Examples for these equations are given in appendix 8.2.

The hydraulic conductivity K(θ) is strongly dependent on the water content. It decreases with decreasing water content. The maximum value is reached at full water saturation. At this point it equals the saturated hydraulic conductivity $K_S$.

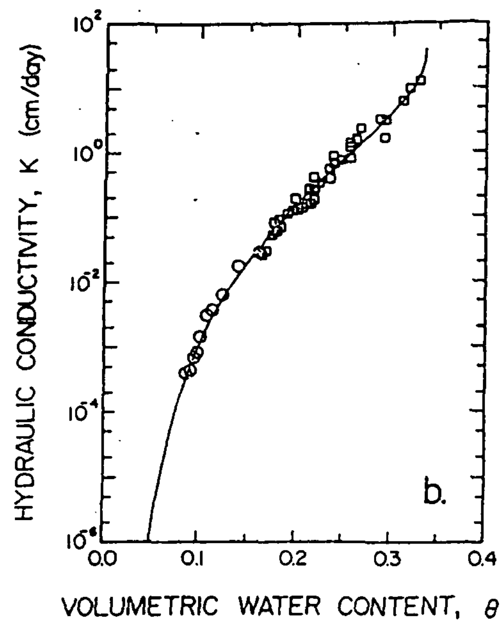



*Figure 3-4* *Hydraulic conductivity versus water content (van Genuchten & Šimunek, 2002)*

Both functions K(θ) and ψ(θ) are dependent on the pore size distribution of the considered media.

The application of Richards equations implies that the water flow is determined by both matrix and gravity potential. However, with increasing pore diameter the impact of the porous matrix on the water flow declines. Thus, gravity is the only driving force for water transport through macropores. Another difference between water flow through fine-pored media and macropores is that water moves much faster through larger pores due less flow resistance ("friction forces"). The phenomenon of rapid water flow along "preferential pathways" was first recognized by Schumacher (1864), who stated that "the permeability of a soil during infiltration is mainly controlled by big pores, in which the water is not held under the influence of capillary forces". Several attempts to describe these flow processes in a mathematical way have been made so far. In order to simplify the mathematical description the network of preferential flow paths is usually lumped into one channel. In the following, the most common approaches for modeling preferential flow through soils respectively fractured rocks are briefly presented. All of them have already been applied to simulate the water flow through sanitary landfills.

- Poiseuille Law (1841)

$$Q = \frac{\pi \cdot r^4}{8\eta} \nabla p$$ **Equation 3-5**

| | |
|---|---|
| $Q$ | *Volume flux [$m^3\ s^{-1}$]* |
| $\eta$ | *Dynamic viscosity [$kg\ m^{-1}\ s^{-1}$]* |
| $r$ | *Radius of the pipe [m]* |
| $\nabla p$ | *Pressure gradient [$kg\ m^{-2}\ s^{-2}$]* |

This equation was developed to calculate steady state, laminar water flow through cylindrical pipes. Uguccioni & Zeiss (1997) applied this equation (implemented in the model PREFLO) to compute preferential water transport through waste columns.

- Kinematic wave assumption (after Beven & Germann, 1981)

$$q = b(\theta_{ma})^a$$ **Equation 3-6**

| | |
|---|---|
| $q$ | *Volume flux density [$m\ s^{-1}$]* |
| $b$ | *Hydraulic conductivity (conductance) under saturation [$m\ s^{-1}$]* |
| $\theta_{ma}$ | *Water content of macropores participating in the flow process [$m^3\ m^{-3}$]* |
| $a$ | *Dimensionless exponent [-]* |

It is assumed that the water flow through macropores follows a power function of the water content. This approach was used by Bendz (1998) and Hartmann (2000) to simulate the water flow through landfills.

- Richards equation for the macroporic flow

In a so called dual-permeability approach (Gerke & van Genuchten, 1993, 1996) the macropores are assigned a higher hydraulic conductivity than the micropores. The water flow is calculated separately for both domains (macropores and micropores). Thus, for each point of the flow field two velocities, two water contents and two hydraulic heads exist. The landfill model of Obermann (1999) is based on this approach.

Although, the importance of macroporic flow for subsurface hydrology is undisputed, at present the application of complex models taking this phenomenon into account is restricted to theoretical investigations and laboratory studies (Šimunek et al., 2003). In particular the large numbers of parameters involved and the current lack of standard experimental techniques to obtain them, inhibit the employment.

The approaches originating from soil physics can only be adapted to landfills, under the condition that flow mechanism and underlying assumptions are applicable for preferential flow in landfills.

# 4. A New Approach for Modeling Leachate Generation

Due to the limitations and drawbacks of existing landfill models a new approach for predicting leachate generation and water storage in MSW landfills is required. The two main principles for this model are: universal applicability (in particular variable boundary conditions at field scale) and incorporation of heterogeneous water flow. Both principles have not been combined so far in existing mathematical landfill reproductions.

In the following chapters a new approach for modeling water flow in MSW landfills is gradually derived. Starting from simple black box considerations (water balances) the determining factors and processes evolved are identified and discussed.

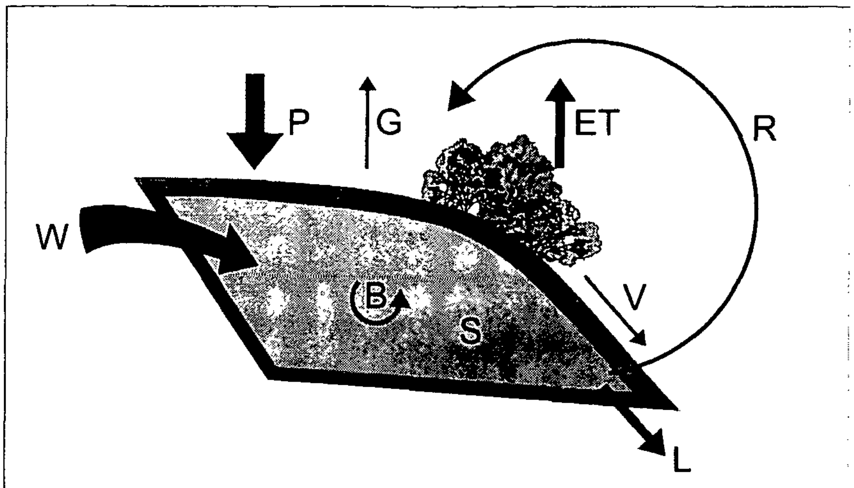
## *4.1. Water balance considerations*

The simplest way to describe the hydrology of landfills is performing a water balance. Thereby only input, output and storage of water into or out of the system are considered. This general concept of modeling is known as system identification technique and the resulting model is called black-box model (Bender, 1978).

The general water balance equation for MSW landfills can be written as follows (modified after Baccini et al., 1987):

$$P + W + R \pm B - (ET + G + L + V) - S = 0 \qquad \textbf{Equation 4-1}$$

Water is introduced into the landfill through the moisture of the landfilled waste material (W), as precipitation (P) and in some landfills by water addition during landfilling and recirculation of leachate (R). Some of the precipitation may run off as overland flow (V), and some may evaporate from the waste material or be removed by transpiration from the vegetation cover (ET). Inside the landfill body some water may be generated or consumed by biochemical processes (B), whereby anaerobic decomposition processes of organic matter consume water and during aerobic decomposition water is produced. The remaining water must accumulate (water storage S) or be discharged by drainage (leachate L). Beside the drainage of leachate water may also leave the landfill as vapor (G) through the gaseous phase. The storage of water against gravity is caused by the texture of the waste material itself (materials with capillarity or water retaining properties) as well as by the texture of the landfill (capillary voids between waste materials).




| | | | |
|---|---|---|---|
| P ... | Precipitation | L ... | Leachate |
| ET ... | Evapotranspiration | G ... | Water output by gas flux |
| V ... | Runoff | B ... | Biochemical water production/consumption |
| W ... | Water input by waste deposition | S ... | Water storage |
| R ... | Water recirculated | | |

J. Fellner, 2004

*Figure 4-1* *Water balance of landfills*

To provide a simple example, water balance investigations into the experimental landfill "Breitenau" (description of the landfill site see chapter 5.1) are briefly presented.

The water input into the landfill Breitenau caused by precipitation (P) was evaluated using mean values of rainfall obtained from three meteorological stations (Neunkirchen, Saubersdorf and Wr. Neustadt) nearby the landfill site (see Figure 5-3). Due to the lack of measurements concerning the initial water content of the landfilled waste (W), this parameter was estimated using literature data. According to Brunner et al. (1983) and Ehrig (1989) an initial water content of MSW of 30 % (WS) was assumed. This figure is in agreement with water content measurements of MSW originating from the same community the landfilled waste came from (Schachermayer et al., 1994). The amount of water added during landfilling, re-circulated leachate (R) and surface runoff (V) were calculated according to data given in Riehl-Herwirsch et al. (1995). Water was added during the landfilling of the waste in order to ensure a high water content, and thus, better conditions for decomposing microorganisms. The rate of potential evapotranspiration was evaluated after Haude (1954) using regionally adapted factors of phenology (Dobesch, 1991) and crop coefficients according to the vegetation cover. In order to obtain the required actual evapotranspiration for the water balance calculations the availability of water in the cover layers had to be considered. Thereto the water flow model LEACHW (Hutson & Wagenet, 1992) was applied. This model enables to compute the actual evapotranspiration in dependency of the available water over the root depth. Due to several assumptions concerning evapotranspiration the determined values are highly uncertain. The leachate discharge (L) was measured using different methods during the considered time period (1987 – 2002). Electromagnetic flow meters, mechanical gauges and differences in the water level of the leachate collection tanks were used to determine the leachate generation rate. The amount of condensate in landfill gas (G) was supposed to be 60 g water per $m^3$ gas (Rettenberger, 1987). The biochemical water production/consumption (±B) was computed according to Pöbel (1964) for aerobic degradation processes, and according to Stegmann & Ehrig (1980) for anaerobic processes. The amount of water stored within the landfill body (S) was estimated by solving Equation 4-1.

The "Breitenau" landfill is divided into three separate compartments with different capping systems (see Figure 5-1). Table 4-1 gives the results of the water balance calculations for all three compartments including the uncertainties. The values are referred to the initial moist mass of the landfilled waste in order to obtain comparable data.

***Table 4-1 Parameters of the water balance (1988-2002) for the landfill Breitenau [l/t MSW]***

| | | Compartment I | Compartment II | Compartment III |
|---|---|---|---|---|
| Landfill cover (surface) | Precipitation (P) | 1,230±20 | 1,060±20 | 1,130±20 |
| | Surface run-off (V) | -63±6 | -26±3 | -15±2 |
| | Evapotranspiration (ET) | -785±80 | -750±75 | -965±50 |
| | *Climatic water balance (P-ET-V)* | *382±82* | *284±78* | *150±54* |
| Landfill | Water added (R) | 51±3 | 66±3 | 112±6 |
| | Initial water content (W) | 300±30 | 300±30 | 300±30 |
| | Water production (+B) | 11 | 11 | 11 |
| | Water consumption (-B) | -15 | -15 | -15 |
| | Vapor in gas (G) | -3 | -3 | -3 |
| | Leachate (L) | -356±18 | -246±12 | -113±6 |
| | Storage (S) (rounded) | 370±90 | 400±85 | 440±60 |

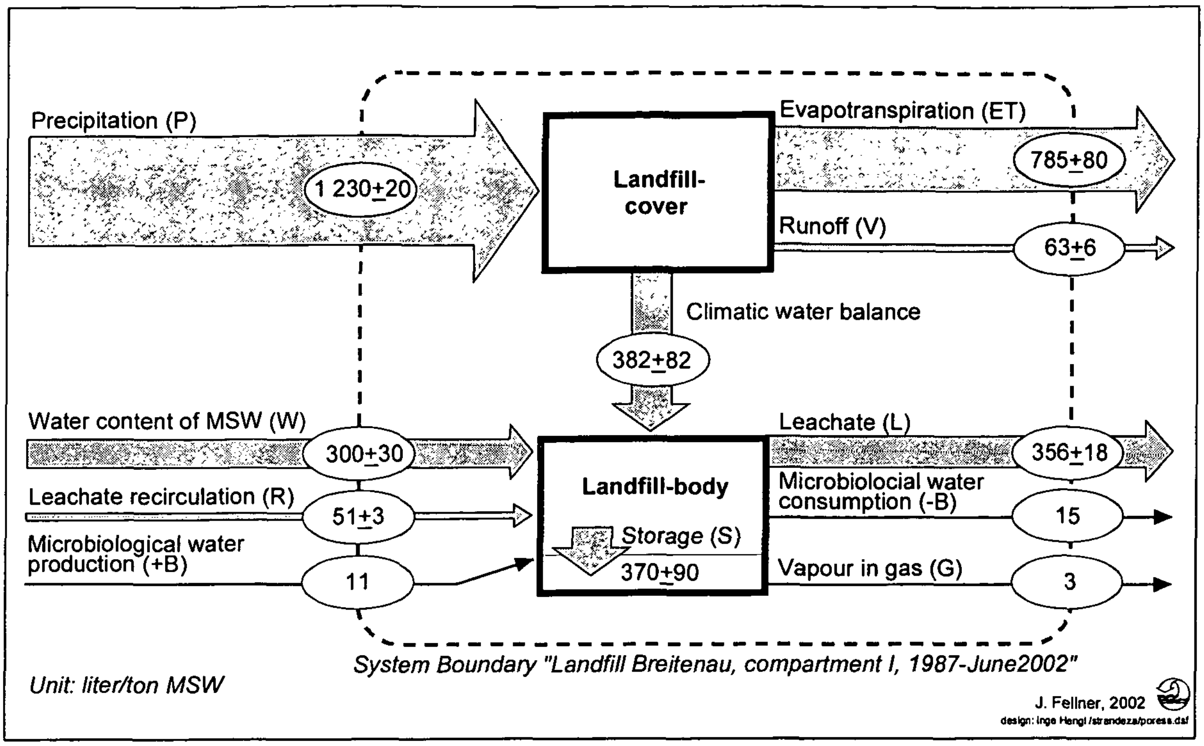



*Figure 4-2 Water balance for the Breitenau landfill – Compartment I*

Precipitation, evapotranspiration and runoff can be summarized to the climatic water balance. This parameter represents the amount of water that percolates from the landfill cover layers into the waste body.

The results indicate that biochemical water production and consumption as well as water losses due to the vapor in landfill gas are non-relevant for the water balance of landfills. Thus, they may be neglected in further considerations. A comparison of the results shows that the water input into the three compartments of the landfill was different during the observation period (15 years). This attributes on the one hand to different capping systems and on the other hand to diverse operation strategies.

Water input expressed as climatic water balance is governed by the capping system and the climatic conditions at the landfill site. This water input turns out to be highest in Compartment I (382 l/t MSW), followed by Compartment II (284 l/t MSW) and Compartment III (150 l/t MSW). Different evapotranspiration is mainly responsible for the diverse water input. The geometry of the compartments (average landfill height) also affects the rate of water coming into contact with waste. Compartment III that is abundantly covered with vegetation shows the highest quantity of evapotranspiration. Plants withdraw more than 85 % of the incoming precipitation.

The second major water input into the landfill beside the climatic water balance is the water content of the landfilled MSW. Furthermore, in the considered case of Breitenau landfill water was added during landfilling, and after landfill closure in the form of re-circulated leachate. For the three compartments differences in the additional water input were reported. At Compartment III recirculation of leachate was carried out till 1995 with a total additional water input 112 l/t MSW. At Compartment I and Compartment II water recirculation was stopped after landfill closure in 1989. Till this date around 51 l/t and 66 l/t MSW have been recirculated at C I and C II, respectively.

Summarizing climatic water balance, irrigation water and initial water content leads to a total water input of around 730 l/t MSW at C I, 650 l/t MSW at C II and 560 l/t MSW at C III. During the same period leachate generation rates at C I of 356 l/t MSW, at C II of 246 l/t MSW and at C III of 113 l/t MSW were registered. This results in an average water storage of around 370, 400 and 440 l/t MSW for the three compartments.

The water balance of the Breitenau landfill is dominated by the following parameters:

- precipitation
- evapotranspiration
- initial water content of the landfilled MSW
- water storage
- leachate
- re-circulated leachate and water added during landfilling
- runoff

Whereby, the relevance of the term re-circulated leachate and water addition during landfilling is specific due to the certain operation strategy at the landfill site Breitenau. Although the importance of other terms is site specific, their general importance can be assumed.

Model approaches for simulating water flow in landfills should incorporate the above mentioned input and output parameters.

Apart from identifying the parameters determining for the water balance of landfills a phenomenon, unaccountable by black-box considerations, can be observed by comparing water entering and exiting the three compartments:

Higher water input into landfills consequentially results in higher leachate discharge, which is not surprising. Higher water input however, does not inevitably increase water storage inside the waste body. For instance, Compartment III with the lowest water input of 560 l/t MSW shows the highest rate of water retention (440 l/t MSW) and Compartment I with the highest water input of 730 l/t MSW shows less storage of water (370 l/t MSW). The transfer coefficients describing the relation between water input, output and storage are significantly diverse for each compartment (see Figure 4-3).

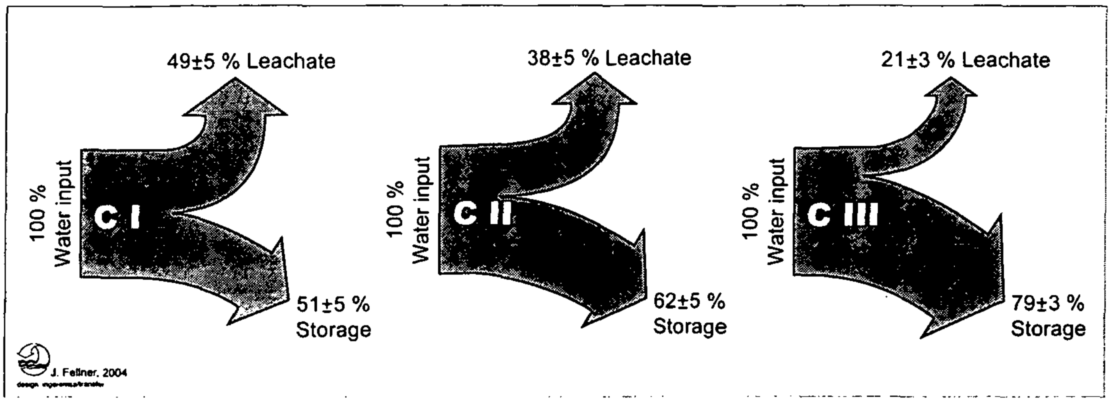



*Figure 4-3* ***Water transfer-coefficients for each compartment (Breitenau landfill)***

Compartment I shows the highest fraction of water release (49±5 %), whereas in Compartment III only 21±3 % of the water input occurs as leachate discharge at the bottom of the landfill. C II lies with 38±5 % of released water in between.

Transfer coefficients for the different water paths depend on the amount of water input, as the storage capacity of a landfill is limited. In a long term view the transfer coefficient for the leachate path must converge to 1. The different transfer coefficients deduced for the three compartments however, may only partly attribute to various water inputs. The absolute figures also indicate least water storage at Compartment I.

Considering that similar waste was landfilled and water input was within the same range, the transfer coefficients at the three compartments should be in the same range. Since similar hydraulic characteristics (storage capacity) of the waste material itself can be assumed. The results of the water balances however, show different hydraulic behavior for each compartment. This fact indicates that the water flow through landfills is not only dependent on the characteristics of the waste material itself.

Low absolute values together with low transfer coefficients for the water storage indicate a highly non-uniform water distribution in landfills. In this case water flow is funneled to less bulk of waste and therefore also the effective storage capacity is restricted to the waste mass participating in water flow. Applying this postulation to the results of the Breitenau landfill would mean that large zones in C I did not get in contact with water yet. Whereas according to the results of C II and C III it can be assumed that dry zones in these compartments are of less importance. Thus, the water flow regime in Compartment II and C III is more uniform compared to C I.

Although comparisons of simple balance considerations may already provide an indication on water distribution in landfills, it is imperative for understanding and studying leachate generation processes to investigate the hydraulic characteristics of landfilled MSW. Obviously, these properties will have a direct impact on the results of any project studying leachate routing just as the hydrologic properties of the subsurface media will affect a groundwater modeling study.

## *4.2. Hydraulic characteristics of municipal solid waste*

Municipal solid waste is due to its origin a highly heterogeneous media. Investigations of Turczynski (1988) showed that the grain size varies from smaller 0.5 mm up to 1000 mm. The hydraulic behavior of single waste components is divergent. It ranges from highly water adsorbent (e.g. paper, textile) to water repellent, from impermeable (e.g. plastic foils) to well permeable materials. Nevertheless, it is possible to determine hydraulic parameters for landfilled waste, if the investigated volume is big enough to be representative for the mixture of materials. Usually

- hydraulic conductivity
- porosity and
- water retention characteristics

are used to characterize the hydraulics of sanitary landfills. In the present chapter an overview of reported hydraulic parameters is given.

### 4.2.1. Hydraulic conductivity

The hydraulic conductivity K represents a measure of the ability of a substance to transmit water. This parameter determines together with the porosity the velocity of a fluid inside the porous media and is therefore important for hydraulic considerations. Figure 4-4 gives a summary of the saturated hydraulic conductivity of MSW from different investigations. Some of the experiments were carried out for different roof pressure resulting in altered waste densities. The reported conductivity values vary between $1\times10^{-2}$ and $5\times10^{-9}$ m/s. The higher values represent waste with a low degree of compaction and measurements performed in-situ by pumping tests, since they contain a larger horizontal component and the conductivity in the horizontal direction is nearly 10 times higher than in the vertical direction (Ramke, 1991; Powrie & Beaven, 1999). With increasing waste density a significant decrease of the hydraulic conductivity is apparent. The rate of the conductivity decline however, is strongly diverse in different experiments.

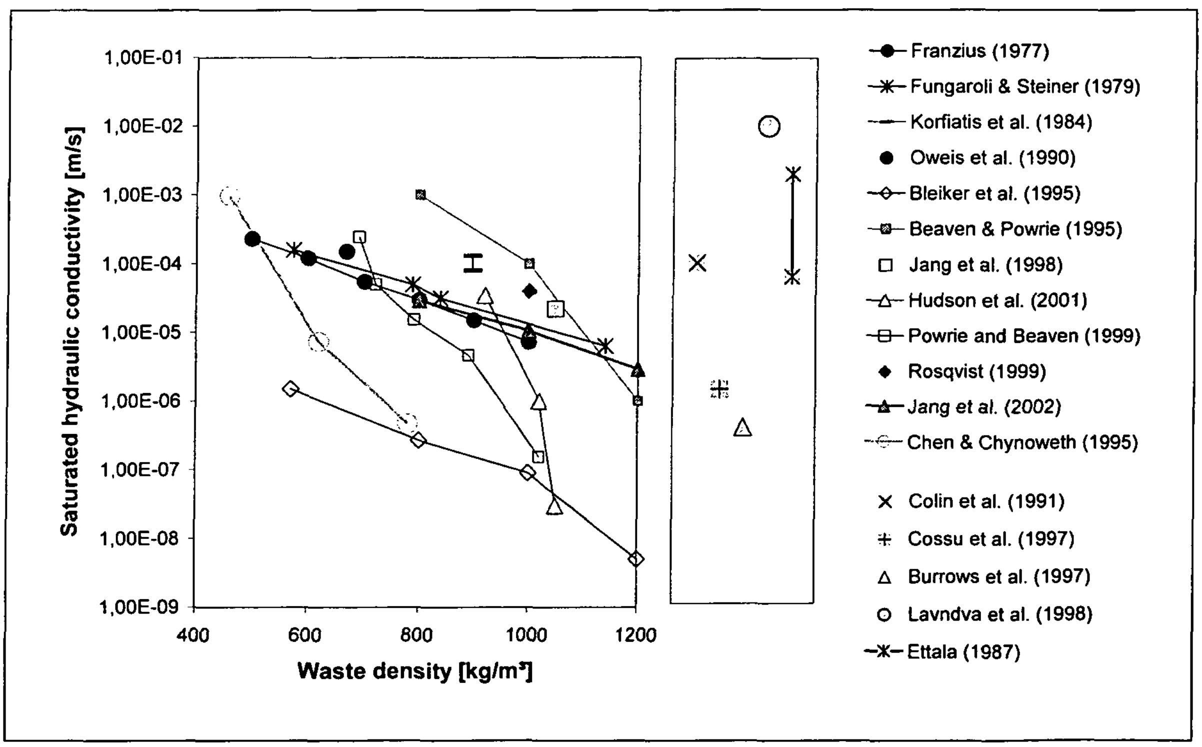



*Figure 4-4 Saturated hydraulic conductivity of MSW*

The saturated hydraulic conductivity $K_s$ represents the ability to transmit water under the condition that all pores are filled with water. Modern sanitary landfills that are operated according to the state of the art (equipped with a leachate collection system at the landfill bottom) are far from water saturation. Thus, water transport within the landfill occurs mainly under unsaturated conditions. It is therefore crucial for water flow investigations to characterize the unsaturated hydraulic conductivity performance of MSW landfills. Hydraulic investigations have primarily been conducted under water saturation. Only a few studies for unsaturated waste were reported in the literature. The results of these studies are summarized in Figure 4-5. Apart from the investigation carried out by Jang et al. (2002) a strong decline of the hydraulic conductivity with increasing matrix potential $\psi_m$ (suction head) is noticed. This means that the permeability of MSW is highly dependent on the water content and the degree of saturation. Low water content is associated with low hydraulic conductivity.

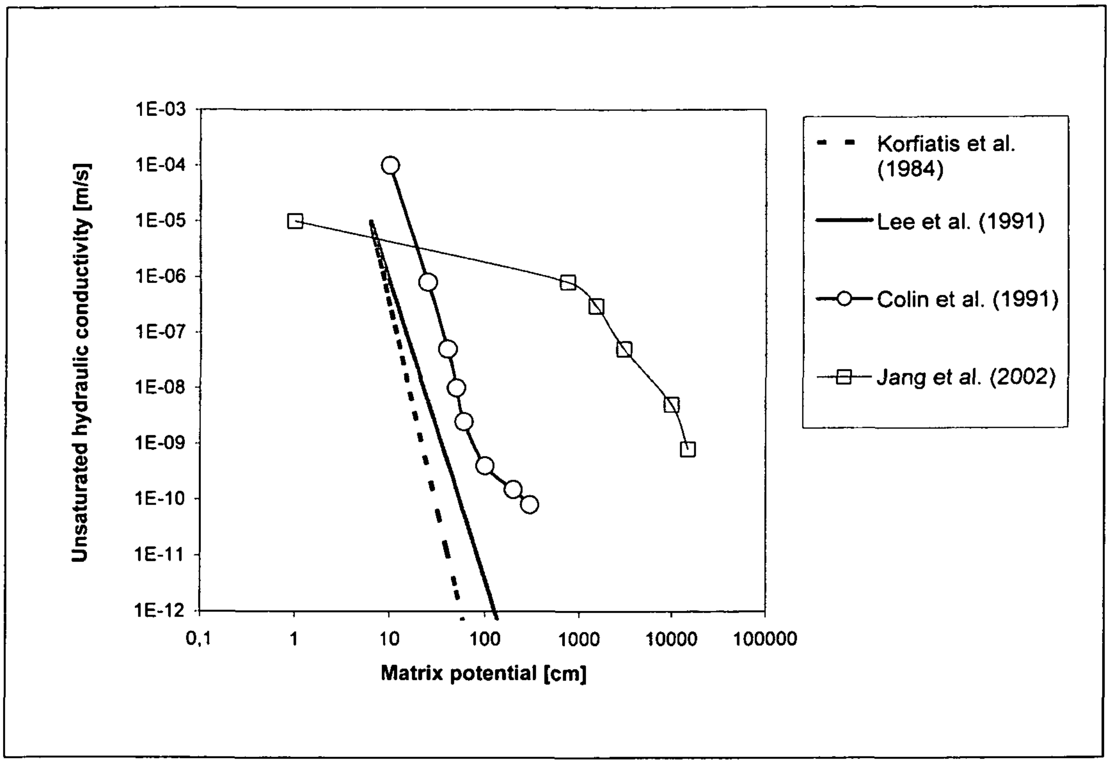



*Figure 4-5 Unsaturated hydraulic conductivity of MSW*

### 4.2.2. Porosity

The porosity n is a measure of the void space in a porous media. It is defined as the ratio between pore volume and total volume. Under saturated conditions the void space is totally occupied by water. The porosity is relevant for hydraulic considerations as it determines the upper limit of the water storage capacity.

In the literature porosity values of MSW landfills between 0.30 and 0.65 were reported (see Table 4-2), whereby the majority of the values lies around 0.50. The porosity of landfilled waste logically decreases with increased compaction energy (Franzius, 1977).

***Table 4-2*** ***Porosity of landfilled MSW reported in the literature***

| Reference | (wet) Density ρ [kg m$^{-3}$] | Porosity n [m$^3$ m$^{-3}$] |
|---|---|---|
| Pacey (1982) | 890 | 0.48 – 0.51 |
| Oweis et al. (1990) | 640 – 1,300 | 0.40 – 0.50 |
| Landva & Clark (1990) | 1,000 – 1,400 | 0.30 – 0.60 |
| Colin et al. (1991) | 400 | 0.65 |
| Zeiss & Major (1993) | 360 – 550 | 0.47 – 0.58 |
| Powrie & Beaven (1999) | 690 – 1,020 | 0.46 – 0.56 |
| Rosqvist (1999) | 1,000 | 0.53 |
| Yuen et al. (2001) | 840 | 0.55 |
| Jang et al. (2002) | 800-1,200 | 0.29 – 0.52 |

### 4.2.3. Water retention characteristics

The water retention characteristics describe the capillarity of a porous media. It is important for the storage of water inside the landfill. The retention characteristics can be expressed using either a single parameter (field capacity) or a defined relationship between water content θ and matrix potential $\psi_m$ (suction head) resulting in the so called water retention curve. The field capacity (FC) gives the maximum amount of water (per volume) that can be retained against gravity force. Typically reported field capacities for MSW landfills range from 0.12 to 0.54 (Table 4-3). Field capacity is basically a function of the waste composition, the density and the porosity. It is expected to change with time, as the degradation of the waste alters its composition (Blight et al., 1992). The time for the water content to increase from its initial value to field capacity can be significant. Bengtsson et al. (1994) found that water was still accumulating in 10-year-old landfills.

*Table 4-3 Field capacity of landfilled MSW reported in the literature (after Yuen et al., 2001)*

| Reference | Field capacity $[m^3\ m^{-3}]$ |
|---|---|
| Remson et al. (1968) | 0.29 |
| Franzius (1977) | 0.16 – 0.45 |
| Holmes (1980) | 0.29 – 0.42 |
| Straub & Lynch (1982°) | 0.30 – 0.40 |
| Korfiatis et al. (1984) | 0.20 – 0.30 |
| Canziani & Cossu (1989) | 0.29 – 0.37 |
| Oweis et al. (1990) | 0.20 – 0.35 |
| Colin et al. (1991) | 0.40 |
| Lee et al. (1991) | 0.32 – 0.54 |
| Zeiss & Major (1993) | 0.12 – 0.14 |
| Schroeder et al. (1994) | 0.29 |
| Bengtsson et al. (1994) | 0.44 |
| Powrie and Beaven (1999) | 0.40 – 0.45 |
| Rosqvist (1999) | 0.41 |
| Yuen et al. (2001) | 0.34 |
| Jang et al. (2002) | 0.26 – 0.45 |

A more detailed characterization of the water holding capability of a porous media is the so called water retention curve. This approach conceptually treats the waste as a bundle of capillary tubes with a range of diameters. The distribution of pore diameters is a material characteristic. It determines the water retention curve that represents the relation between water content $\theta$ and matrix potential $\psi_m$ (or suction head). Several empirical equations have been developed so far to describe this relationship for soils (Brooks & Corey, 1964; Mualem, 1976; van Genuchten 1980; Hutson & Cass, 1987; Vogel & Cislerova; 1988). In order to depict the retention characteristics of waste, parameters from soil models have been simply adjusted. Figure 4-6 gives on overview of the reported water retention curves for MSW. The results vary over a range of two magnitudes (in matrix potential). The strong distinctions are mainly due to different waste composition, compaction, observation scale and measurement method. Additionally wetting and drying cycles have an impact on the retention characteristics (Zeiss & Major, 1993). Despite the large variations at least a general shape of the water retention curves for MSW can be derived from Figure 4-6.

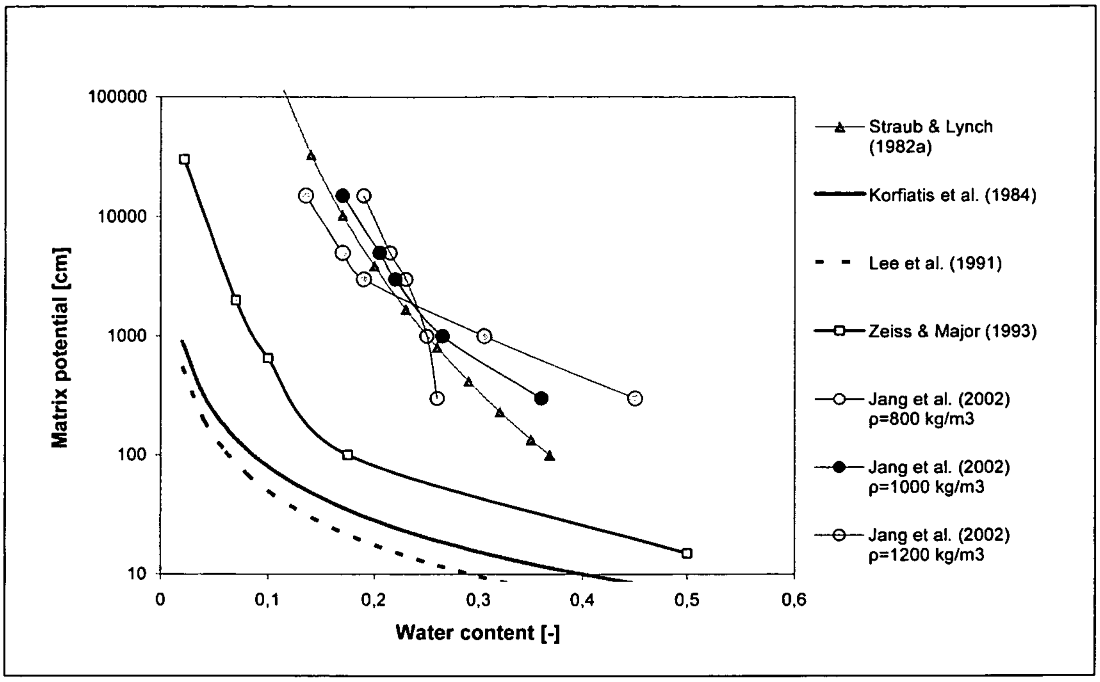



*Figure 4-6 Water retention characteristics of MSW*

All the above mentioned parameters depend on various factors. Waste composition, the degree and way of compaction, the deposition age, the degradation state, the observation scale and also the measurement method govern the hydraulic characteristics of MSW. Therefore, parameters reported in the literature describing the hydraulic properties show large variations. Consequently, a literature review can only give a feasible range of the parameter values.

## *4.3. Leachate hydrographs*

The discharge hydrographs of MSW landfills apparently contain information about the water transport through landfills analogous to the information provided by hydrographs from rivers on the characteristics of their catchment's basin (Holtan & Overton, 1963; Ogunkoya & Jenkins, 1993). Delayed time, shape and amplitude of leachate discharge peaks induced by heavy rainfall events allow at least first qualitative statements about water flow and water storage in landfills.

Leachate hydrographs of different landfills (Franzius, 1977; Ehrig, 1978; Jourdan, 1981; Åkesson & Nilsson, 1997; Döberl et al., 2002; Garcia de Cortázar et al., 2002) all show a quick response after heavy precipitation (that exceeds the retention capacity of the cover layers). Furthermore, a low discharge during dry periods without water input (evapotranspiration exceeds precipitation) is noticeable (see Figure 4-7, Figure 4-8 and Figure 4-9). This certain characteristic of leachate hydrographs is explained by opposed hydraulic properties of solid waste landfills. On the one hand water is retained and slowly released by capillary forces acting in micro-pores within the waste. On the other hand rapidly downward water flow occurs in connected macro-pores and fissures, which are caused by the coarse grading of MSW or differences in landfill settlement. The phenomenon of preferential flow is additionally concentrated by construction elements with high permeability such as gas wells. Furthermore, zones with less compaction or boundary zones represent favored areas for rapid water flow.

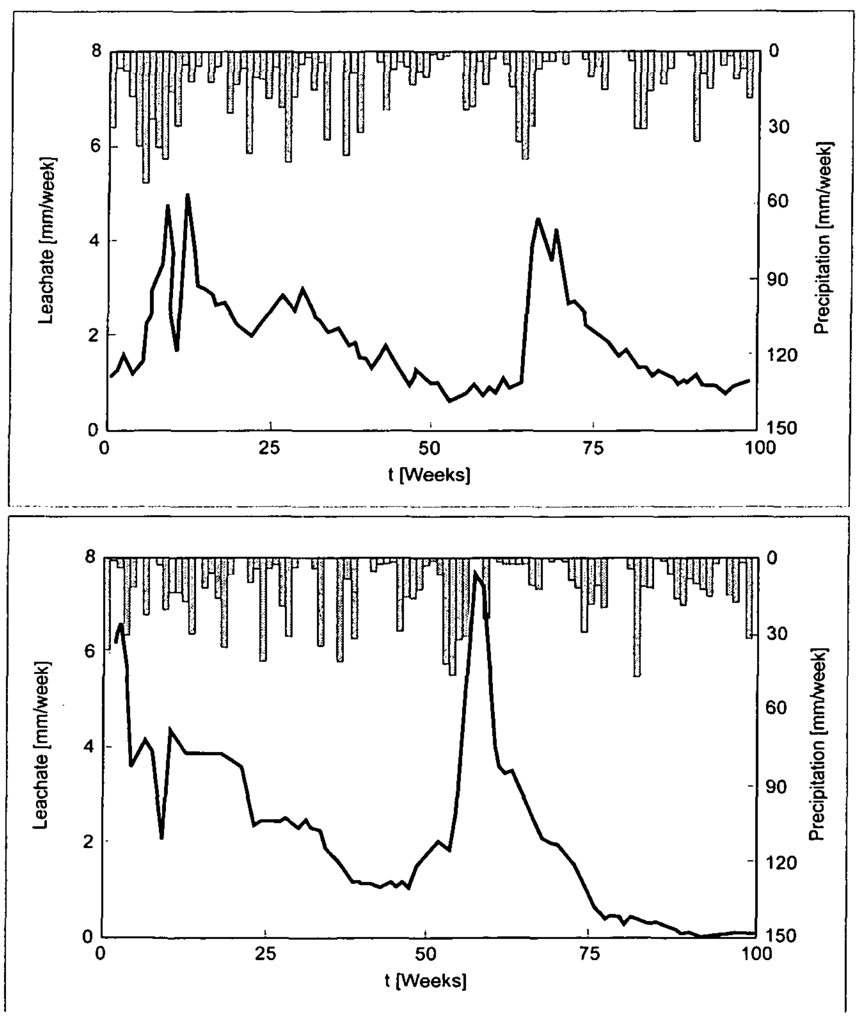



*Figure 4-7 Leachate hydrographs of German landfills (Ehrig, 1978)*

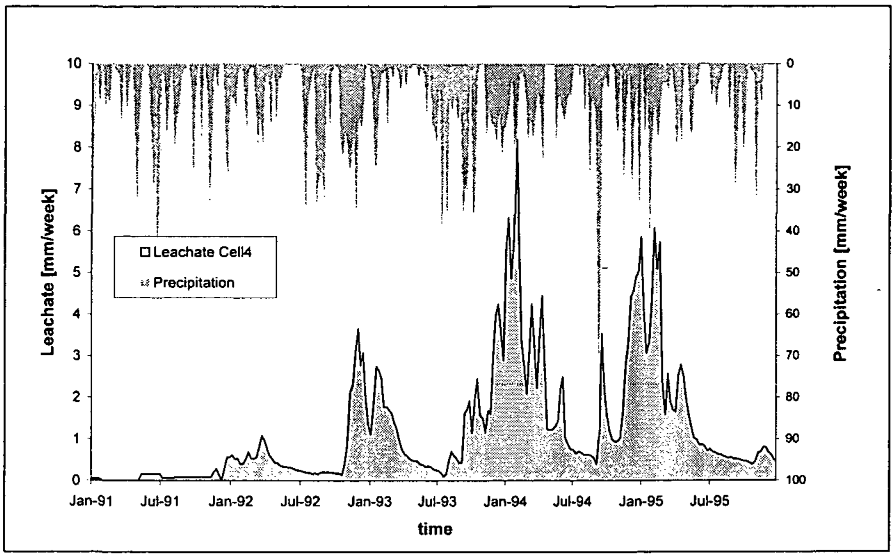



*Figure 4-8* *Leachate hydrograph - Spillepeng test Cell 4 (Åkesson & Nilsson, 1997)*

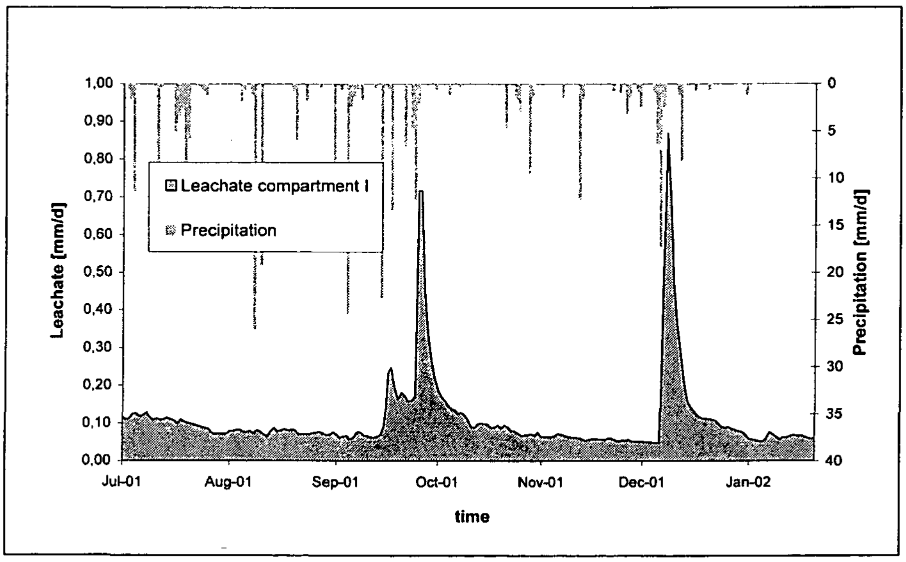



*Figure 4-9* *Leachate hydrograph - Breitenau landfill Compartment I*

## *4.4. Tracer experiments*

In the field of hydrology tracer experiments are conducted to investigate the flow of water through certain systems (e.g. aquifers, soils, karst formations). Thereto the transport media water is charged with a tracer substance at a defined point (feeding point) and in another point (sampling point) the concentration of the tracer substance is recorded over the time. The direct outcome of a tracer test is the so called breakthrough curve (BTC), a chart containing tracer concentration versus time (Figure 4-10). The breakthrough curve enables to characterize the hydraulics of the investigated system.

At sanitary landfills only a few tracer experiments were reported in the literature (Maloszewski et al. 1995; Baumann & Schneider, 1998; Bendz & Singh, 1999; Beaven et al., 2001; Johnson et al., 1998; Rosqvist et al., 2001; Döberl et al., 2002).

Maloszewski et al. (1995) studied the water transport through waste lysimeters and found that up to 40% of heavy precipitation events drains off directly, i.e. within few weeks. Baumann & Schneider (1998), who investigated the water flow through a full sized landfill, came to a similar result. They noticed that 1/8 to 3/8 of the water input reaches the leachate collection system at the landfill bottom with negligible delay. Extensive analysis (Bendz & Singh, 1999; Rosqvist & Destouni, 2000; Rosqvist et al., 2001; Fourie et al., 2001) of tracer substances within waste bodies of different size (0.14 – 545 m$^3$) resulted in the conclusion that only a small fraction of water stored inside the waste takes part in the transport of solutes. Rosquist & Destouni (2000) calculated for an experimental landfill with an average height of 4 meter a mean residence time for the tracer substance of 20 days. The total amount of recovered tracer was around 34 %. The water content actively participating in the transport processes was quantified to be in a range of 6 to 12 %. Figure 4-10 gives an example of breakthrough curves for tracer tests that were performed at small waste columns. The shape is positively skewed with a long right hand tail, indicating a non-uniform transport of the solute through the waste mass. The early peak is attributed to rapid solute transport in favored flow paths (macro-pores) and the prolonged tails indicate slow water flow in less mobile domain (micro-pores).

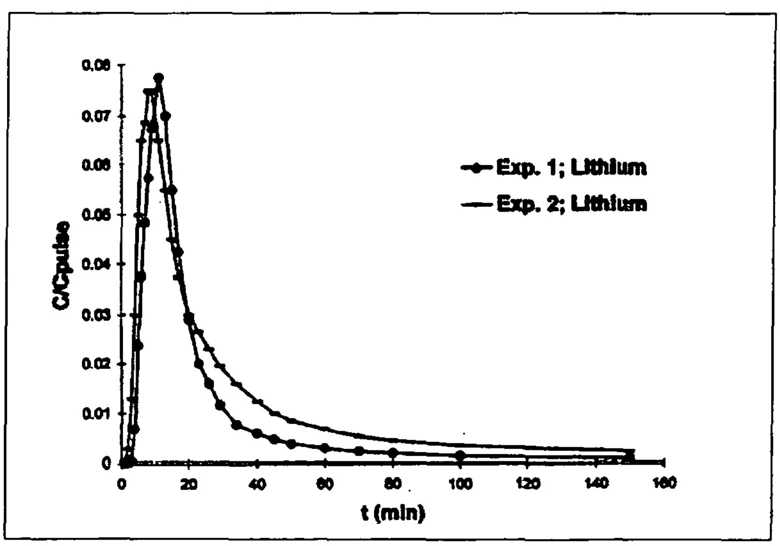



***Figure 4-10 Breakthrough curve of tracer tests (Rosqvist & Bendz, 1999)***

Beaven et al. (2001) performed a tracer test at a full-scale landfill, whereby the experiment was carried out under fully saturated conditions and the main flow direction of water was horizontally. This was due to the fact that no leachate collection system existed at the landfill bottom. The investigations indicate a drainable porosity taking part at the solute transport of around 3 %.

Results of Johnson et al. (1998) showed that even in more homogeneous landfills of bottom ash (compared to MSW) preferential flow paths play an important role for the hydrology.

Döberl et al. (2002) calculated for the Breitenau landfill (Compartment I with 12 m height) a mean tracer resistance time of 90 days and a water content participating in transport of 0,08 %. The recovery rate of the tracer substance however, was less than 20 % for this experiment (observation period: 260 days).

The phenomenon of rapid tracer breakthrough was noticed in all studies (e.g. Table 4-4) and attribute to preferential flow paths prevailing in landfills. The given breakthrough times expressed in bed volumes indicate the period till first significant appearance of tracer. One bed volume represents the total amount of water stored inside the landfill. Assuming completely uniform-flow conditions the breakthrough time of the tracer would be one bed volume.

***Table 4-4 Breakthrough times of tracer tests reported in the literature***

| Reference | Landfill volume [$m^3$] | Landfill height [m] | Breakthrough time [bed volume] |
|---|---|---|---|
| Rosqvist et al. (2001) | 0.14 | 0.65 | ~ 0.68 – 0.87 |
| Bendz & Singh (1999) | ~ 3.5 | 1.2 | ~ 0.05 – 0.1 |
| Rosqvist & Destouni (2000) | ~ 545 | 4 | ~ 0.22 |
| Beaven et al. (2001) | ~ 15,000 | 20 | ~ 0.011[#] |
| Baumann et al. (1998) | ~ 50,000 | 6 | ~ 0.006 |
| Döberl et al. (2002) | ~ 30,000 | 12 | ~ 0.0008 |

[#] *Predominately horizontal flow under saturated conditions*

The data of Table 4-4 shows that the water flow becomes more non-uniform with rising volume and height of the landfill. That is demonstrated by declining breakthrough times. The assertion that the water transport in landfills is more non-uniform with rising depth was confirmed by Rosqvist et al. (1997), who noticed acceleration in the solute velocity towards the bottom of an experimental landfill. Taking the reduced hydraulic conductivity in greater depths into account (e.g. Bleiker et al., 1995; Powrie & Beaven, 1999) an increase in the solute velocity must attribute to heightened preferential flow. This finding is contrary to assertions of different studies (e.g. Bendz, 1998; Zeiss, 1997) that assume analogous to soil a more uniform water flow in greater depths.

## *4.5. Water flow pattern and its implication for water flow modeling*

The importance of channel flow for the hydrology of landfills have been identified by several researches (e.g. Ham & Bookter, 1982; Ehrig; 1983; Zeiss & Major, 1993) and already partly implemented into concepts for modeling water routing in landfills (Young & Davis, 1992; Uguccioni & Zeiss, 1997; Bendz, 1998; Obermann, 1999). As described in chapter 3.1.1.4 the waste mass in these models is split into a channel domain with rapid water flow surrounded by a matrix domain with slow water movement. Although the mathematical approaches describing the water transport in the two domains are diverse, all developed concepts are derived from the framework that has been carried out for non-uniform water flow in soils. Compared to soils, landfills are more heterogeneous, which results in a bigger fraction of preferential flow. This fact was taken into account by adjusting decisive parameters. The underlying assumption, the similarity between water flow in landfills and soils has not been justified or even discussed yet. The following section will point out the flow pattern in landfills on a macroscopic scale and will compare it with the non-uniform water flow in cracked or fissured soils.

As mentioned above landfilled MSW (due to its origin and its composition) is a highly heterogeneous media. The hydraulic behavior of single waste components ranges from highly water adsorbent to water repellent, from impermeable to well permeable materials. Nevertheless, it is possible to determine hydraulic parameters (chapter 4.2) for landfilled waste, if the investigated volume is big enough to be representative for the mixture of materials.

Considering the whole landfill body in contrast to the waste material itself, the degree of heterogeneity is increasing and therefore also representative volumes (Bear, 1972). This fact implicates that parameters determined at a small scale are invalid to characterize the whole landfill mass. In particular construction elements (e.g. gas wells or daily cover layers), areas with low mechanical compaction and boundary zones are responsible for the enhanced heterogeneity. Furthermore, landfilling and compaction of waste in thin layers leads to a horizontal stratification within the landfill. Consequently, the hydraulic conductivity shows a distinctly anisotropic behavior. Powrie & Beaven (1999) observed 10 times higher conductivity in the horizontal direction. Thus, a major part of the water flow through landfills occurs horizontally (Burrows et al., 1997). Additionally, the anisotropic behavior is increased due to the horizontal orientation of impermeable materials such as plastic sheets. Water is retained above these impervious surfaces inside the landfill body. Hanging water tables in different depths inside landfills, which have been reported in several investigations (e.g. Stegmann, 1990; Riehl-Herwirsch et al., 1995), attribute to those barriers caused by compaction and impermeable sheets, respectively. The retained water is forced to continue its flow in a more or less horizontal direction. Vertical channels and fissures resulting from the heterogeneous nature of the waste itself, from differences in landfill settlements, and from vertical construction elements with high permeability, short the horizontal pathways, and enable fast downward water transport inside landfills. The impervious surfaces (e.g. straightened plastic sheets) serve as water suppliers for the preferential flow paths. The described mechanism, water retaining and horizontal flow towards vertical channels, is repeated within every waste layer. This leads inevitably to a funneling of water in preferential flow paths with increasing depth. Subsequently, water flow becomes more non-uniform towards the landfill bottom. This predication is in agreement with investigations carried out by Wiemer (1982), Gabr & Valero (1995), Öman et al., (1999) and Yuen et al., (2001), who noted higher spatial differences in the water content towards the landfill bottom. Ziehmann et al. (2003), who studied the spatial difference in water supply of leachate collection systems, confirmed the existence of a highly non-uniform water flow at the landfill bottom.

Figure 4-11 provides a schematic for the water flow pattern inside MSW landfills. The picture is adopted from Mesu (1982), who first pointed out the importance of impermeable layers for the water movement. He compared the water transport inside landfills with the water flow from roofs.

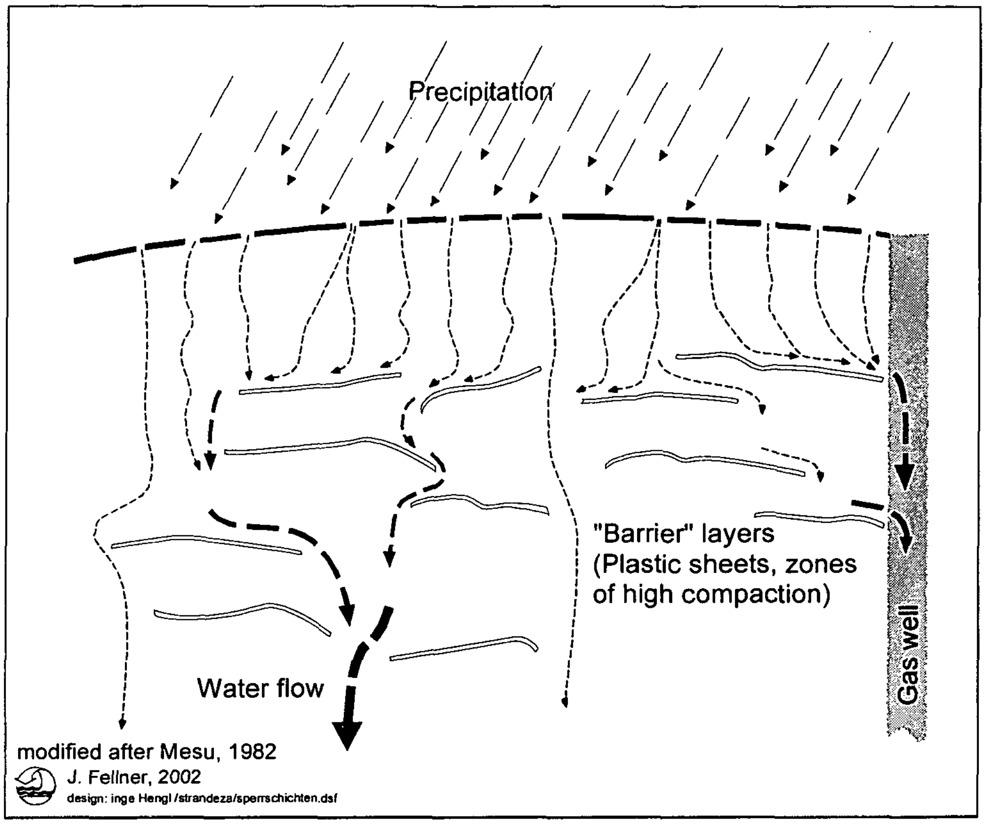



*Figure 4-11 Water flow pattern in MSW landfills (modified after Mesu, 1982)*

The general water flow pattern in landfills is mainly determined by the structure of the landfill (e.g. impermeable horizontal surfaces, vertical channels). The portion of channel flow however and the matrix flow is not only dependent on the structure itself but also on the water application rate. New flow channels may develop or can be reached due to higher backwater above the impermeable surfaces during periods with high infiltration rates (Jasper et al., 1985). Even during dry periods (no additional water input) water is retained above plastic "barriers" and supplies the preferential flow paths.

Heterogeneous (fissured) soils, which are usually used as a comparable media for MSW landfills, show a different hydraulic behavior (Figure 4-12). No vertical flow limitation comparable to impermeable sheets is found in soils. Thus, water flow mainly occurs in the vertical direction. The application rate of water plays an important role for the water movement and its distribution in soils (Germann & DiPietro, 1996). During dry spells water transport is limited to the soil matrix only, whereas under wet conditions (during or short time after water infiltration events) macro-pores and fissures also contribute to the downward water movement. The soil matrix sorbs some of the water bypassing in preferential pathways, since capillary forces are acting in micro-pores. The fraction of water infiltrated from the fissures into the matrix depends on the water content of the soil matrix.

The mechanism and the degree of preferential flow in landfills and soils are strongly diverse. Thus, the distribution and pattern of preferential flow is different. Investigations in soils (e.g. Bundt et al., 2000) show that the effect of favored flow paths is becoming minor with depth, while the non-uniformity of the water flow in landfills increases towards the bottom.

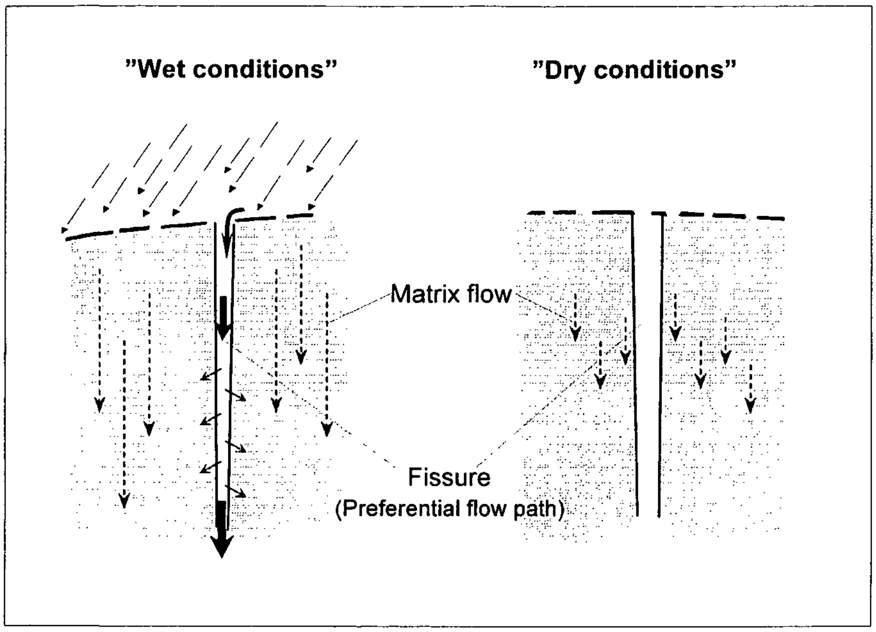



*Figure 4-12 Water flow in cracked soils during wetting periods and dry spells*

The main differences in the water flow between landfills and soils can be summarized as follows:

- in landfills the flow pattern mainly depends on the structure of the waste, whereas in soils the application rate of water is a decisive parameter determining the water flow
- contrary to soils, in landfills a large amount of water flow occurs in horizontal direction due to the anisotropic characteristic of landfills
- preferential flow occurs in soils only during wetting periods, while landfills show significant channel flow also during dry periods
- in soils the direction of water flow between the two domains is more or less restricted to flow from the channels into the fine pored matrix, while both flow directions are possible in landfills

- in soils the heterogeneity of the water flow decreases with depth, contrary to landfills where an increase of preferential flow in greater depths is observable
- finally the degree of non-uniformity of water flow in landfills is bigger than in soils

Due to the differences in the hydrology of landfills and soils, it can be concluded that present model concepts are based on inadequate assumptions. In particular the distinctive horizontal water flow in landfills (see Figure 4-11) caused by impermeable layers makes any abstraction into a one-dimensional vertical flow model insufficient. Even if a two-domain concept is realized, a major characteristic is neglected.

Based on conceptual considerations a simplification of the illustrated flow pattern (Figure 4-11) was proposed (see Figure 4-13). Analogous to previous modeling concepts (Uggucioni & Zeiss, 1997; Bendz, 1998; Obermann, 1999; Hartmann, 2000) the flow field is divided into a matrix domain with slow water transport and a channel domain with rapid water flow. Contrary to previous concepts however, the two-domain flow field is implemented in two dimensions. The matrix zone is characterized by low permeability and high water retention capacity, while the vertical channel domain shows high hydraulic conductivity and low (or even no) retention capacity. Thus, the matrix acts as storage zone for water and enables water release during dry spells. The channel domain allows fast downward water flow through the landfill and is responsible for the quick response of leachate discharge after precipitation.

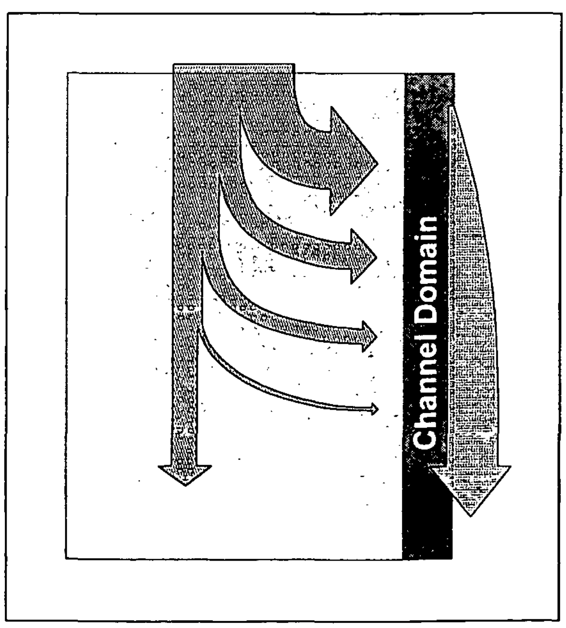



***Figure 4-13 Simplified water flow pattern in MSW landfills***

The spatial extent of the matrix zone is predominant. However, the channel domain is effective for the water transport. Near the surface water mainly flows in the matrix domain and the water distribution is more or less uniform. With increasing depth water is drained from the matrix domain into the channel, whereby the amount of water transferred from the matrix to the channel decreases with depth. At the landfill bottom a main fraction of water is originated from the preferential flow path. In consequence, the bulk of waste, which is by-passed by channel flow, increases towards the bottom.

## *4.6. Implementation of the new modeling concept into an existing software tool*

In principle the implementation of the flow pattern (displayed in Figure 4-13) into a mathematical transport model could be accomplished by two different options (Figure 4-14).

- The first manifest option (Figure 4-14 left side) to reproduce the proposed flow pattern is to introduce horizontal layers with low permeability. These layers retain water and enable the transport of water into the channel domain. However, quantitative information about their extent, permeability and inclination is lacking. Therefore modeling attempts would require several assumptions, and thus, extensive parameter calibration due to numerous unknown factors (hydraulic characteristics of matrix and channel domain as well as for each "barrier", spatial distribution, extent and inclination of "barrier" layers, and anisotropy of the hydraulic conductivity). A practical application of this option seems to be unworkable.
- The other possibility to ensure water flow from the fine-pored matrix domain into the preferential flow path according to the proposed flow scheme can be accomplished by defining the channel domain as a suction pipe, whereby the suction head is increasing towards the top of the landfill (Figure 4-14 right side). Higher suction head causes heightened water movement into the channel domain. The proposed concept reduces unknown characteristics of the impermeable layers to the suction head in the preferential flow path and the anisotropy of the hydraulic conductivity of the matrix domain.

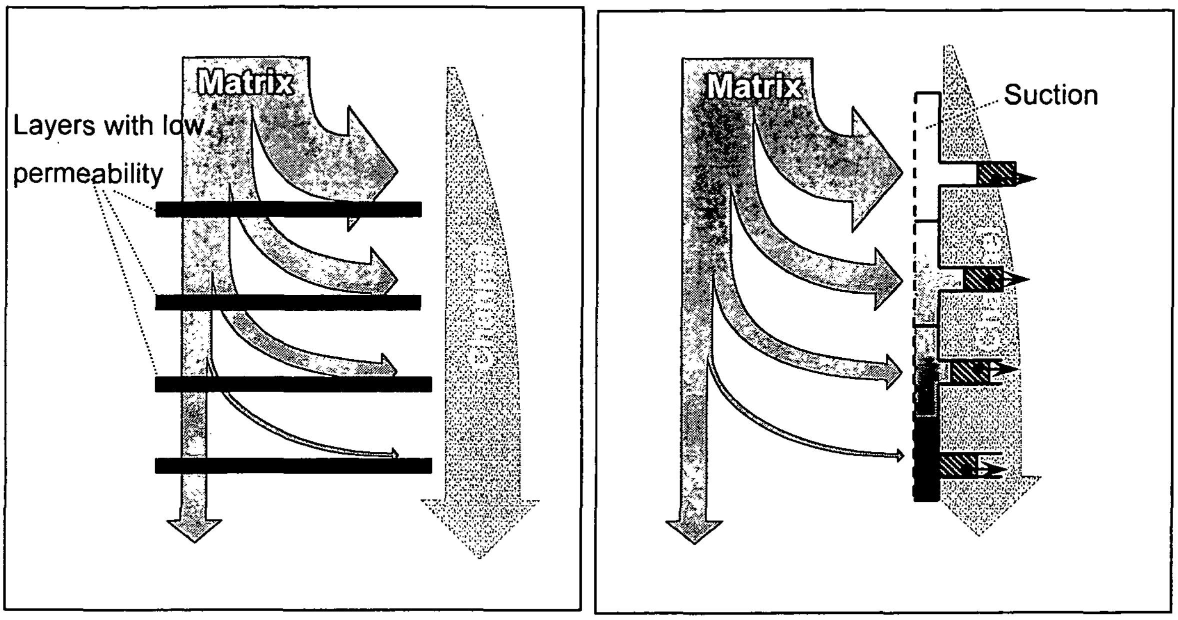



*Figure 4-14 Modeling options to implement the assumed flow pattern*

Due to the less unknown parameters and simpler practicability, the two-dimensional two-domain concept assuming suction power inside the channel is preferred. The uniformity respectively non-uniformity of the water flow can be controlled by the "properties" of matrix and channel domain.

The implementation of this option into an existing mathematical model for simulating water transport requires a tool including the following processes and abilities:

- two-dimensional simulation of water flow in variably saturated porous media
- spatial differences and anisotropy of the hydraulic characteristics within the two-dimensional flow field
- controllable suction power acting inside the preferential flow path

In addition appropriate software for simulating water flow through landfills must incorporate:

- the boundary conditions: seepage (at the bottom) and variable flux (at the top)
- water losses due to evaporation, evapotranspiration and surface runoff

Although numerous simulation tools (e.g. MODFLOW, Hill et al., 2000; HST3D, Kipp, 1997) of varying degree of complexity have been developed so far to quantify water flow in porous media, most programs were designed for groundwater flow, and thus, consider saturated conditions only. Two dimensional water transport under variably saturated

conditions is considered in e.g. SUTRA (Voss, 1984), VS2DI (Healy & Ronan, 1996), SEEP2D (GMS, 1996), FEMWATER (Lin et al., 1997), HYDRUS-2D (Šimunek et al., 1996). A comparison of these models shows that the above mentioned requirements for the two-dimensional two-domain concept are best fulfilled by the program HYDRUS-2D. It is the only model that enables a varying suction power (acting inside the channel domain) by applying the method of linear scaling of hydraulic properties (Vogel et al., 1991). HYDRUS-2D was developed at the U.S. Salinity Laboratory in Riverside to simulate water flow, solute and heat transport in variably saturated porous media at various boundary conditions. A short description of the software and its field of application are given in appendix 8.1.
The water flow in HYDRUS-2D is calculated using Richards equation (1931). This formula implies that gravity and capillary forces govern water flow. An assumption that concurs with the situation in the matrix domain quite well, but it is not applicative for the flow conditions in the channel domain. In fact it is contrary that, on the one hand large pores enable a rapid water flow and on the other hand capillary forces should dominate the water movement in the channel domain. However, this physical "error" is even required in order to realize the assumed flow pattern (Figure 4-14 right side), because water can only drain from the matrix if the capillary potential in the channel domain is lower. The introduction of capillarity acting in the preferential flow path represents from a physical point of view a suction pipe for the matrix domain, which enables that water is draining from the matrix domain into the channel.

Figure 4-15 (left side) shows the graphical operators interface of HYDRUS-2D with a defined two-dimensional two-domain flow field. Ascertaining hydraulic properties of matrix and channel domain as discussed above (high retention capacity and low permeability for the matrix domain and low retention capacity and high permeability for the channel domain) results in the given pressure (suction) head distribution (Figure 4-15 right side), a snap shot out of a simulation over a longer period. The dark colored zones indicate low suction head, whereas bright colors refer to areas with high suction head. The resulting pressure (suction) head distribution revealed by schematically implemented streamlines confirms that the new model concept enables to reproduce the investigated flow pattern.

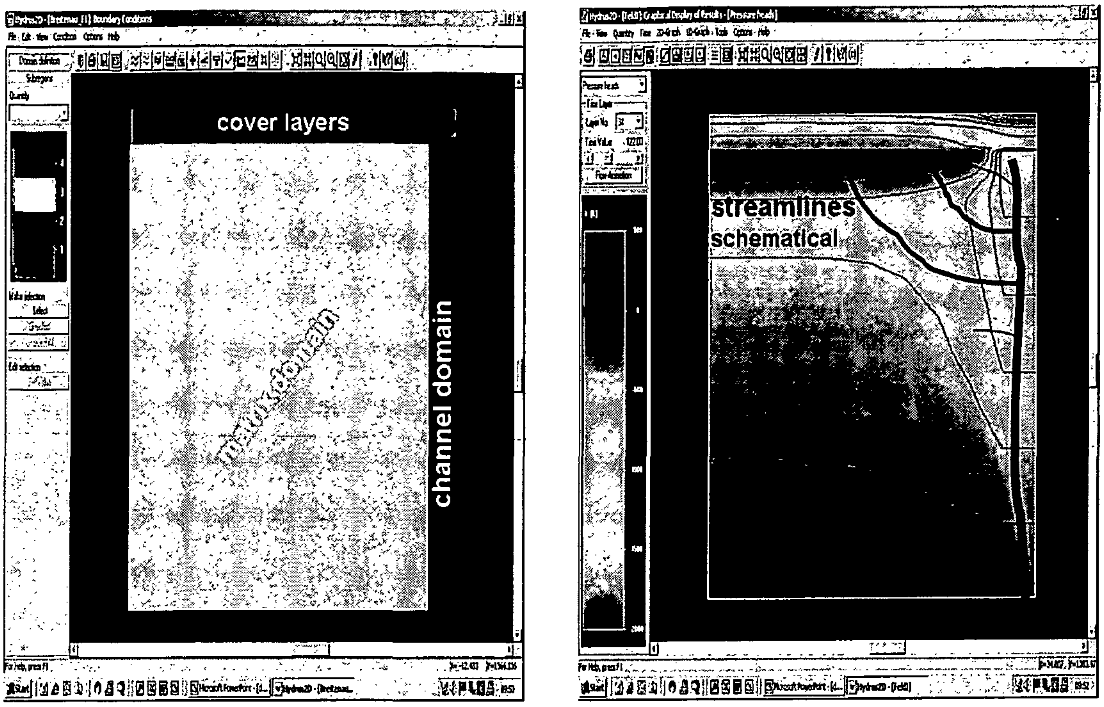



*Figure 4-15 Two-dimensional two-domain concept (left side) and resulting flow field (right side)*

## 4.7. Required input information of the new model

The introduced two-dimensional two-domain water flow model based on HYDRUS-2D requires the following input information:

- "meteorological data":
    - precipitation
    - potential evapotranspiration
    - surface runoff
    - information about the vegetation cover
- flow field definition:
    - partitioning of matrix domain and channel domain
    - suction head (channel domain)
- hydraulic characteristics of the landfilled waste (matrix and channel domain) and the cover layers:
    - saturated/unsaturated hydraulic conductivity, anisotropy
    - water retention characteristics (van Genuchten model)
- initial conditions:
    - water content of the waste and the cover layers

### 4.7.1. Meteorological data

The meteorological data at the landfill site are required on a daily basis, whereby for HYDRUS-2D the potential evapotranspiration must be split up into evaporation and transpiration. This can be accomplished using information about the crop cover. Allen et al. (1998) proposes a linear relationship between crop cover fraction and the portioning into evaporation and transpiration (Figure 4-16).

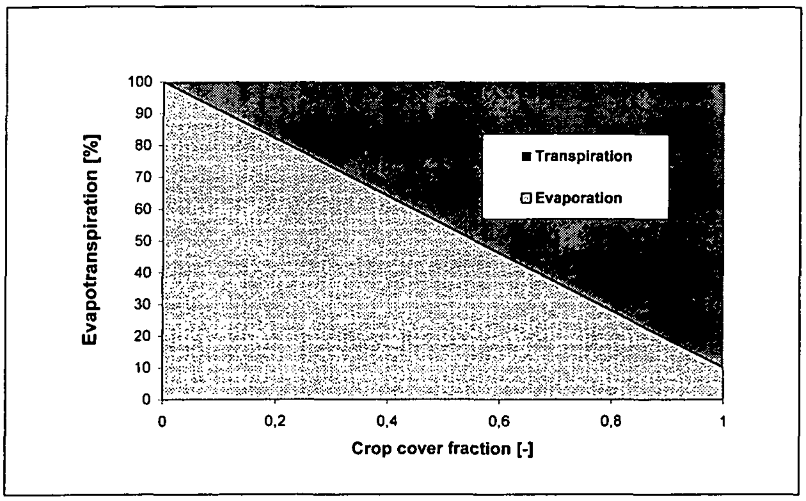



*Figure 4-16 Subdivision of evapotranspiration in dependence of the crop cover fraction*

In addition to the crop cover fraction, information about the root depth and the root distribution is needed to reproduce the hydrologic system landfill cover in a mathematical form.

### 4.7.2. Flow field definition

As described above, the proposed two-dimensional two-domain approach for modeling water flow in landfills postulates a separation of the waste mass into matrix and channel domain, whereby the area of the matrix zone is predominant. The proportion of the flow field is assumed as follows: 97 % matrix domain and 3 % channel domain. This partitioning is based on tracer experiments of Rosqvist (1999) and Beaven et al. (2001), who stated that the spatial fraction of preferential flow paths on the whole landfill body is less than 5 %. Additionally to the partitioning of the domains, the dimensions of the considered profile are of importance. For the simulations landfill profiles with less than 1 m width were assumed. The limiting to

1 m is based on two factors: On the one hand the limitation is necessary to curtail calculation times and on the other hand the limitation is based on a physical cause. Waste lenses inside the landfill reach a maximum horizontal length of 1 m (Bendz, 1998), which implies that preferential pathways can occur in this distance. In order to ensure alike proportions of the flow field, the width of the simulated landfill profile is altered with the landfill height. For a landfill of 10 to 12 m height a profile width of 1 m was chosen. This width reduces to 0.5 m for a landfill of 5 to 6 m height only.

One parameter introduced to accomplish and affect the presumed flow pattern is the scaling factor $\alpha_h$ of the pressure (suction) head (Vogel et al., 1991) in the channel domain. It allows confining the water flow pattern within a particular range.

### 4.7.3. Hydraulic characteristics of waste and cover layers

The hydraulic characteristics of cover layers can either be determined taking samples and performing laboratory test or using previously reported parameter values for similar soils.

In contrast, hydraulic parameters of MSW in particular of the two-domains (matrix and channel) are difficult to obtain. This is mainly due to two reasons. First, the two-domains are conceptual materials for the model approach rather than real existing media. The matrix domain however, can be understood as undisturbed waste mass. The second reason is that large representative volumes are necessary to characterize the mixture of materials. Therefore, hydraulic tests must be conducted at large samples. Furthermore, in previous studies reported parameters of MSW are hardly adjuvant. The values vary considerably (see chapter 4.2). Thus, reported hydraulic parameters provide only a first estimate for the parameter values of the matrix domain. The actual determination of the parameters must be performed during the calibration of the flow model.

For the channel domain a qualitative estimation of the hydraulic characteristics results in high hydraulic conductivity $K_s$ and low porosity n respectively saturated water content $\theta_s$.

Table 4-5 gives feasible ranges of different hydraulic parameters. A robust physical background of these values is missing, as these parameters are representing not only the media characteristics themselves but additionally interactions of the hydraulic system.

*Table 4-5* ***Feasible ranges of the hydraulic parameters of the two-domains using the van Genuchten model (1980)***

| "Material" | | Residual water cont. $\theta_r$ [-] | Saturated water cont. $\theta_s$ [-] | Form coefficient $\alpha$ [1/m] | Form coefficient $n_g$ [-] | Saturated conductivity $K_s$ [m/s] | Pore-connectivity l [-] |
|---|---|---|---|---|---|---|---|
| Matrix domain | min | 0 | 0.35 | 0.5 | 1.1 | $5\times10^{-8}$ | 2.0 |
| | max | 0.2 | 0.55 | 5.0 | 1.6 | $5\times10^{-6}$ | 40 |
| Channel domain | min | 0 | 0.005 | 0.5 | 1.2 | $1\times10^{-4}$ | 0.1 |
| | max | 0.001 | 0.05 | 5.0 | 3 | $1\times10^{-2}$ | 5.0 |

### 4.7.4. Initial conditions

Initial conditions required for the simulations with HYDRUS-2D are limited to the initial water content of the different "materials" (cover layers, matrix, and channel domain). The exact knowledge of the initial water content of the cover layers is less important, as no significant water storage over a longer time period (years) is possible within theses layers. In contrast the situation for the landfill body, in particular for the matrix domain storage processes over long periods are highly relevant. Therefore, a good estimation of the initial water content is crucial. Initial water contents vary less compared to other hydraulic properties of MSW. Table 4-6 gives an overview of published values. The majority of the reported mass water content values lie between 25 and 35 % (referred to wet mass of MSW).

*Table 4-6* ***Initial mass water content of MSW***

| Reference | Initial mass water content $m_w$ [m%] WS* |
|---|---|
| EAWAG (1975, cited in Brunner, 1976) | 30 |
| Spillmann & Collins (1986) | 26 |
| Baccini et al. (1987) | 30 |
| Ehrig (1989) | 30 |
| Reimann & Hämmerli (1995) | 12 – 35 |
| Schachermayer et al. (1994) | 27 – 30 |
| Fehringer et al. (1997) | 30 |
| Morf et al. (2003) | ~22 |
| Skutan & Brunner (2003) | 37.5 |

* *WS ... referred to wet mass*

In order to obtain the required input data for HYDRUS-2D the values must be converted from mass's percentage $m_w$ to volumetric water content $\theta$. Therefore, information on the density $\rho$ of the landfilled waste is required.

$$\theta = \rho \cdot m_w \qquad \textbf{Equation 4-2}$$

| | |
|---|---|
| $\theta$ | *Volumetric water content [$m^3\ m^{-3}$]* |
| $\rho$ | *Wet density of the landfilled waste [$Mg\ m^{-3}$]* |
| $m_w$ | *Water content, referred to wet mass [$m^3\ Mg^{-1}$]* |

As mentioned above the water flow model must be calibrated and validated using data on the leachate discharge. The temporal resolution of theses data must be at least on a weekly basis. If only monthly values are available the model calibration and validation would be limited to water balance considerations only, as internal water flow processes can not be determined at this time scale. Calibration and subsequent validation of the flow model must be carried out using different time series of leachate discharge. The chronology of the data sets used is irrelevant. In order to ensure reliable calibration and validation, applied leachate records must show alternations in discharge.

The following parameters are predominantly determining for the introduced two-dimensional two-domain flow model and need to be adjusted during the calibration procedure.

- hydraulic characteristics of matrix domain and preferential flow path
- anisotropy, ratio between horizontal and vertical conductivity of the matrix domain
- scaling factor for the pressure (suction) head of the preferential flow path

Additionally the hydraulic parameters of the cover layers must be trimmed within a plausible range.

# 5. Simulation Results (Model Calibration and Validation)

The proposed two-dimensional two-domain model for calculating water flow through MSW landfills was validated using data from two landfill sites. For the first application data from the experimental landfill Breitenau (Riehl-Herwirsch et al., 1995) situated in Lower Austria was available. Observed leachate generation rates from this site were used to calibrate and subsequently validate the introduced model concept. The second calibration and validation of the model was conducted during a research visit at Lund University using information from landfill test cells at the Spillepeng site in Malmö, Sweden (Nilsson et al., 1992).
Both landfills are characterized by a field scale size and a good scientific documentation of operation parameters over a longer period.

## *5.1. Case study experimental landfill Breitenau*

### 5.1.1. Site description

The experimental landfill Breitenau is located at a former gravel mining site 60 km south of Vienna. Since it is one of the best documented landfills (Riehl-Herwirsch et al., 1995; Binner et al., 1997; Döberl et al., 2002), it represents a unique opportunity to investigate the behavior of organic "reactor" landfills. The site was filled up with around 95,000 tons of MSW in the years 1987 and 1988. The landfill is divided into three separate compartments of different size, with different capping systems and different base liner systems (see Figure 5-1). Compartment I (C I) is covered with a thin gravel layer (0.2 m) and above a silt layer of around 0.9 m. Approximately 35,000 tons of MSW have been landfilled in this compartment. Compartment II (C II) contains 25,600 tons of waste. The cover of this field is not uniform. Half of C II is only covered by gravel (1.3 m), whereas the other half has an additional layer of compost (0.7 m). At Compartment III (C III) the same capping system (gravel and compost) as at the second half of Compartment II was installed. The landfilled waste at C III amounts 33,200 tons. Mineral dams and geomembranes separate the three compartments. The base liner system of C I and C II consists of a mineral liner (1.4 m silt) in combination with a geomembrane for control measurements. At Compartment III a sealing made up by a geomembrane was installed.

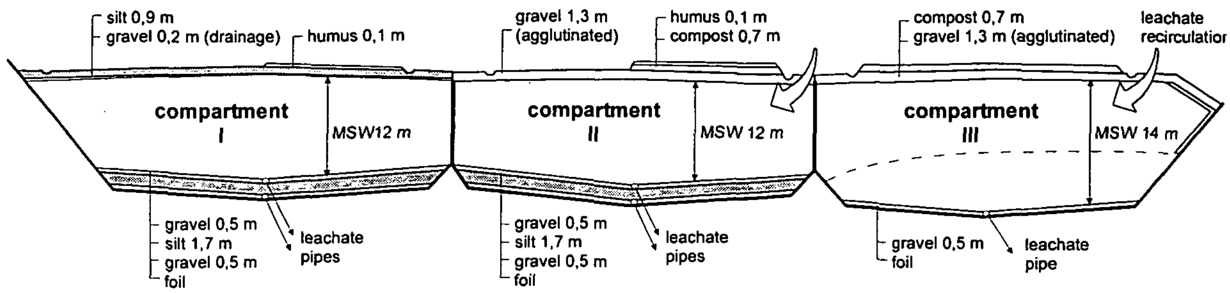



*Figure 5-1 Cross-section of the landfill Breitenau (Huber et al., 2004)*

Drainage layers of coarse gravel were placed above the base lining systems. These layers are connected to drain pipes which lead to the leachate collection house at the landfill base. In the collection house, the discharged water is collected for each compartment separately in tanks and then pumped to the public sewage system. Since the operation of the landfill in 1987, the leachate discharge has been measured using different methods resulting in data of different temporal resolution. In the first years after landfilling (till July 1997) differences in the water level of the leachate collection tanks were used to determine the outflow. From July 1997 to June of 2001 leachate generation was measured using mechanical gauges. Finally in July 2001 the discharge registration method was upgraded to electromagnetic flowmeters in combination with data loggers. These devices enable to record leachate outflow with high temporal resolution (e.g. registration interval of ten minutes).

The purpose of the initial project "Hausmüllversuchsanlage Breitenau" (Riehl-Herwirsch et al., 1995) was to evaluate the influence of waste filled gravel pits on the groundwater. In particular the suitability of sludge derived from gravel washing plants as a barrier between groundwater and waste was investigated. Additionally, the influence of different cover layers on the water balance was studied. It was shown that the capping system built up by compost and gravel (Compartment III) results in least leachate. Whereas the mineral top sealing system with silt (Compartment I) failed after two years, because an obvious increasing of leachate discharge after heavy precipitation events was noticed.

### 5.1.2. Input information

As stated in section 4.7 the introduced model requires information on the hydraulic characteristics of the cover layers and the waste domains (matrix and channel domain), and the meteorological conditions prevailing at the landfill site. In the following a brief summary of the required input data for the modeling effort at the landfill Breitenau is given.

#### 5.1.2.1. Meteorological data

- Precipitation:

The required input precipitation was derived using mean values of three meteorological stations (Neunkirchen, Saubersdorf and Wr. Neustadt) nearby the landfill site (Figure 5-3).

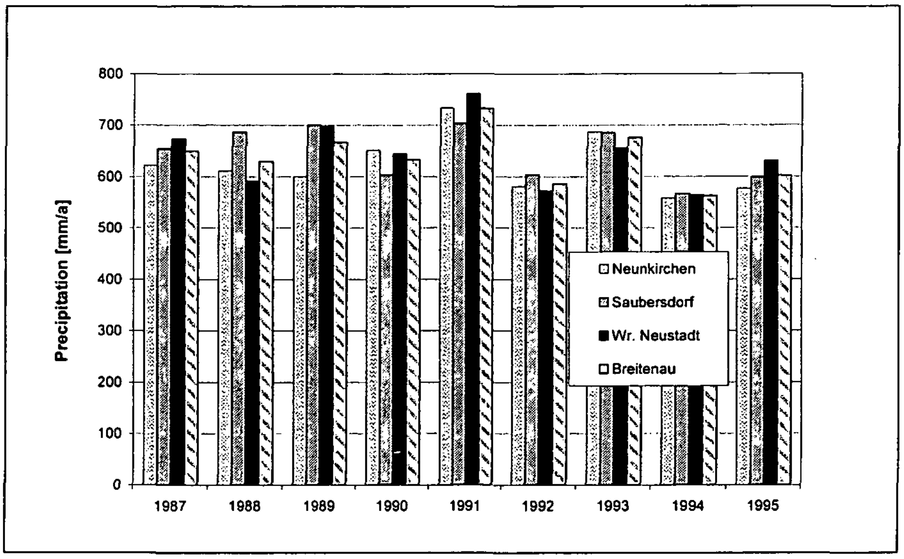



*Figure 5-2* *Annual precipitation of the meteorological stations nearby the landfill Breitenau*

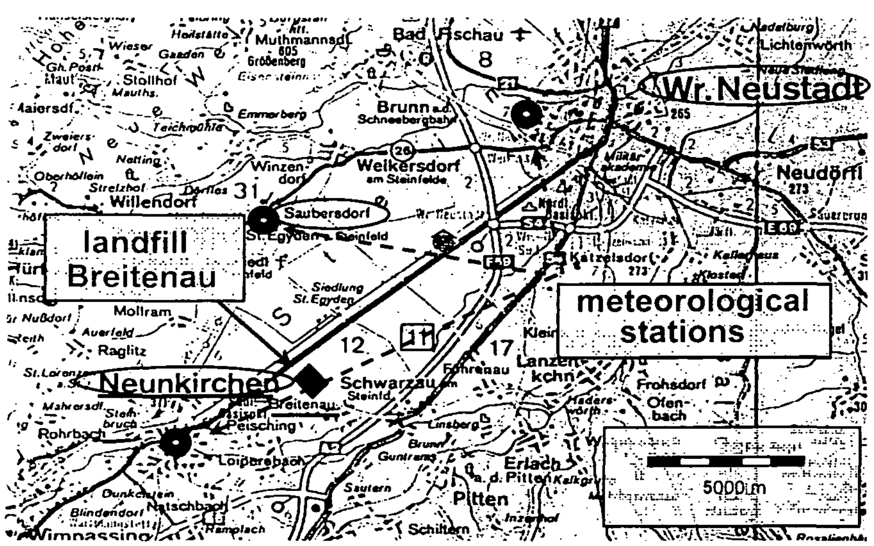



*Figure 5-3* *Location of the landfill Breitenau and meteorological stations nearby*

- Evapotranspiration:

The rate of the reference evapotranspiration $ET_0$ at the landfill site was evaluated according to Haude (1954) using regionally adapted factors of phenology (Dobesch, 1991). The reference values represent the potential evapotranspiration $ET_p$ of grass of 12 cm height during the growing season. In order to obtain the potential evapotranspiration of a different vegetation cover, the reference values $ET_0$ are multiplied by a crop coefficient $K_c$. This coefficient is specific to the crop type and its developmental stage. For vegetation covers of numerous different crops an approximately linear relationship between crop coefficient $K_c$ and the crop cover fraction can be assumed (see Table 5-1 according to Allen et al., 1998).

***Table 5-1*** ***Relation between crop cover fraction and crop coefficient***

| Crop cover fraction [%] | Crop coefficient $K_c$ [-] |
|---|---|
| 100 | 0.95 – 1.15 |
| 75 | 0.75 – 0.85 |
| 50 | 0.55 – 0.65 |
| 25 | 0.4 – 0.5 |

Beyond the growing season and for bare soils, a dependency of the crop coefficient $K_c$ from the reference evapotranspiration $ET_0$ and the rainfall interval according to Allen et al. (1998) is assumed (Figure 5-4). The average interval between significant rainfall events was estimated to 7 days at the landfill Breitenau.

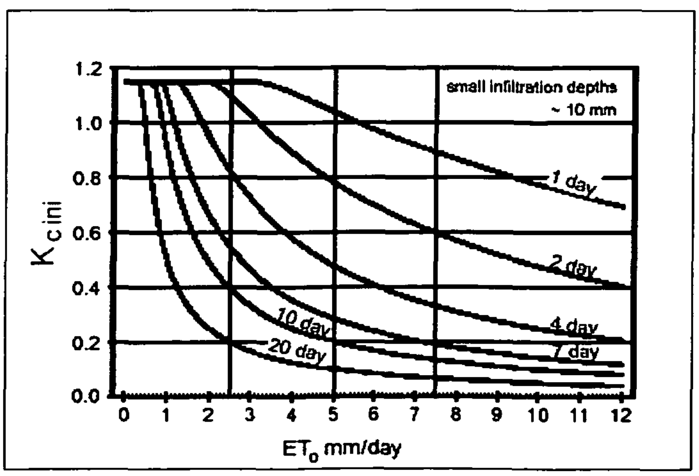



***Figure 5-4*** ***Relation between crop coefficient, reference evapotranspiration and rainfall interval (Allen et al., 1998)***

The required input parameters (air temperature and moisture content at 2 p.m.) for the calculation after Haude (1954) were derived from the meteorological station Wiener Neustadt, as it is the only station nearby measuring these parameters. Additionally to the computation after Haude (1954), evapotranspiration was estimated using the common method of Penman-Monteith (Bevan, 1979). Due to the lack of data on wind speed and sunshine hours (required for the calculation after Penman-Montheith) at the landfill site, feasible ranges for these parameters had to be assessed. Figure 5-5 gives a comparison of annual evapotranspiration values calculated according to Haude and Penman-Monteith assuming an average wind speed of 1 to 2 m/s (Riehl-Herwisch et al., 1995) and a relative sunshine duration of 0.42 (ZAMG, 2002). The evapotranspiration values determined with different methods match within the domain of uncertainty (caused by estimations regarding wind speed and sunshine hours).

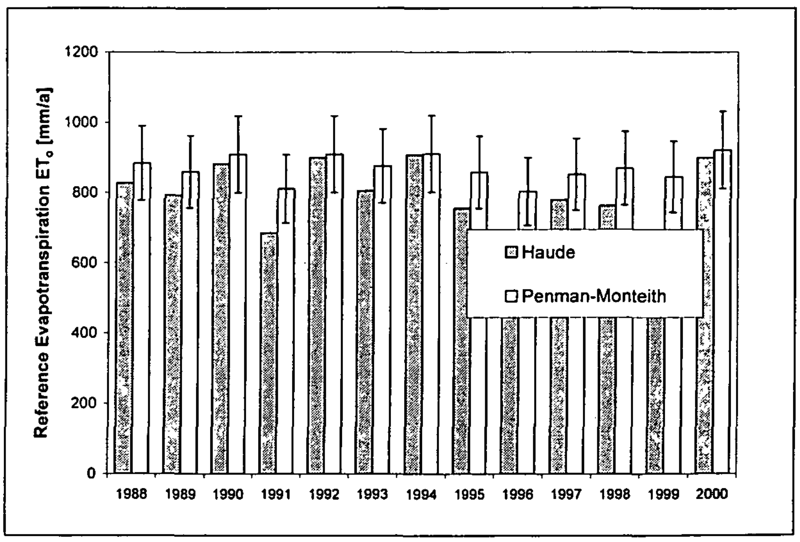



***Figure 5-5 Comparison of annual reference evapotranspiration (after Haude and Penman-Monteith)***

The method after Haude was preferred to determine the reference evapotranspiration at the landfill site, since some of the parameters required for the Penman-Monteith equation are not available or only in form of annual averages. Small deviations between the results (Figure 5-5) of both methods show that this approach is adequate.

Information on the runoff from the landfill exists only till December 1991 in the form of monthly values. Settlements of the landfill surface led to changes in the general slope direction (lowest point of the surface in the center of the compartments), which made overland flow out of the compartments impossible.

As mentioned above, the generation of leachate was measured at different intervals. Since the calibration and validation of the model requires data with high temporal resolution, only observed discharge values since June 2001 are practical for this purpose.

### 5.1.2.2. Hydraulic properties

Additionally to meteorological data the landfill model requires information on the hydraulic properties of the cover layers and the landfilled waste (matrix domain and channel domain). The cover materials of the Breitenau landfill were characterized by the Institute of Hydraulics and Rural Water Management at the University of Natural Resources and Applied Life Sciences, Vienna (Loiskandl, 2001). Saturated hydraulic conductivity, grain size distribution, bulk density and porosity of the different materials have been determined. A summary of the results is presented in Table 5-2 and Table 5-3.

***Table 5-2 Hydraulic conductivity, porosity and density of the landfill cover layers (Breitenau)***

| Cover material | Saturated hydraulic conductivity k [cm/d] | Porosity n [-] | Bulk density $\rho_d$ [Mg/m³] |
|---|---|---|---|
| „Silt" | 90 – 250 | 0.25 – 0.37 | 1.66 – 2.05 |
| „Gavel" *agglutinated* | 2 – 40 | 0.27 – 0.29 | 1.98 |
| "Gravel" *drainage layer* | 500 – 1,500 | – | – |
| „Compost" | 200 – 550 | 0.66 – 0.77 | 0.53 – 0.78 |

***Table 5-3 Grain size distribution of the landfill cover layers (Breitenau)***

| Cover material | Coarse grit (> 2mm) | Fine grit (< 2mm) | Sand (<2mm) | Silt (<63µm) | Clay (<2µm) |
|---|---|---|---|---|---|
| | [Bulk-%] | | | | |
| „Silt" | 56 – 64 | 36 – 44 | 40 – 42 | 40 – 43 | 15 – 19 |
| „Gravel" *agglutinated* | 74 – 82 | 18 – 26 | 55 – 59 | 27 – 33 | 12 – 14 |
| „Compost" | 51 – 66 | 34 – 49 | 62 – 71 | 23 – 30 | 4 – 8 |

By applying so called pedo-transfer-functions PTF, information about the grain size distribution was combined with the porosity and the bulk density to estimate the water retention characteristics of the different materials. In particular the software Rosetta (Schaap et al., 2001) was applied to assess probable parameter values for the retention model of van Genuchten (1980) which is used in HYDRUS-2D. This model consists of empirical equations that describe the relationship between water content and pressure head (equations see Appendix 8.2). The results of Rosetta, possible ranges of van Genuchten parameters for the different cover layers, are presented in Table 5-4.

***Table 5-4 Ranges of van Genuchten parameters of the landfill cover materials (after Rosetta)***

| Cover material | Method | van Genuchten parameter | | | |
|---|---|---|---|---|---|
| | | Residual water content $\theta_r$ [-] | Saturated water content $\theta_s$ [-] | Parameter $\alpha$ [1/m] | Exponent $n_g$ [-] |
| „silt" | PTF (Rosetta) | 0,03 – 0,05 | 0,23 – 0,34 | 1,5 – 4,2 | 1,16 – 1,38 |
| „Gravel" *agglutinated* | PTF (Rosetta) | 0,03 – 0,04 | 0,22 – 0,27 | 4 – 7 | 1,10 – 1,66 |
| „Compost" | PTF (Rosetta) | 0,04 – 0,07 | 0,50 – 0,65 | 1,5 – 4 | 1,26 – 1,46 |

The "definite" hydraulic properties of the waste domains (matrix and channel) need to be determined by calibrating the landfill model, whereby the parameters are varied within the ranges given in Table 4-5. Additionally, the parameters characterizing the cover layers (Table 5-4) are adjusted during calibration.

### 5.1.3. Results of Compartment I

The model was calibrated using leachate data from June 2001 till December 2001. This period was chosen in order to perform the calibration procedure at two peaks of leachate discharge that were induced by precipitation events. The different vegetation cover and thus different potential evapotranspiration within Compartment I was accounted for using an average value of evapotranspiration. The maximum depth of crop roots was set to 25 cm.

Figure 5-6 presents the simplifications of the hydraulic system for the modeling effort. To avoid unrealistic capillary water rise from the waste mass up into the landfill cover, simulations were conducted separately for the landfill cover and the waste body. Results (seepage) obtained from the system cover layers served as water input for the hydraulic system landfill that consists of matrix domain and preferential flow path.

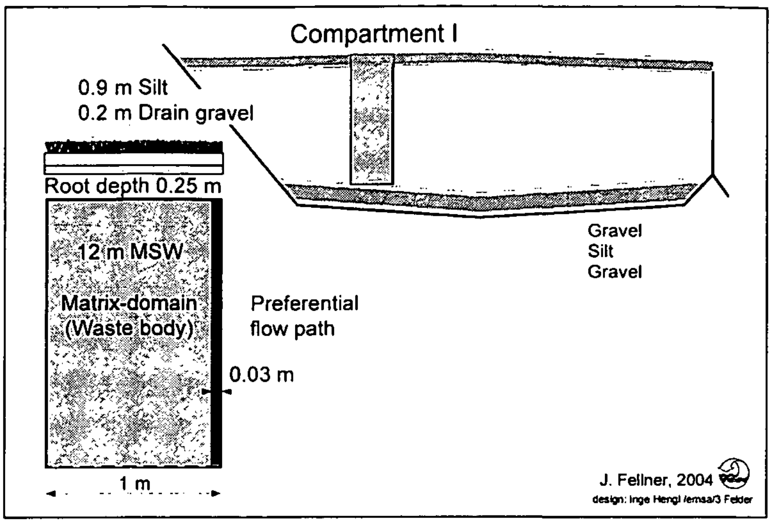



*Figure 5-6* *Simplified model system (Compartment I)*

The model was calibrated using the method of trial and error. The match between simulated and observed leachate discharge was predominantly evaluated by graphical comparison. Finally a quantitative quality grade according to the Gaussian sum of square error (Hartung et al., 1993) was determined.

Applying the two-dimensional two-domain concept, it was possible to predict base flow during dry periods (unaffected by precipitation) as well as discharge peaks after heavy rainfall. The calibrated hydraulic parameters of the cover layers and the waste domains are presented in Table 5-5 and Table 5-6.

*Table 5-5* *Water retention parameters for Compartment I (Breitenau)*

| Material | Water content | | Retention | |
|---|---|---|---|---|
| | Residual $\theta_r$ [-] | Saturated $\theta_s$ [-] | Coefficient $\alpha$ [1/cm] | Exponent $n_g$ [-] |
| Silt | 0.06 | 0.22 | 0.015 | 1.18 |
| Gravel (drain layer) | 0.04 | 0.15 | 0.145 | 1.5 |
| Matrix-domain | 0.15 | 0.5 | 0.02 | 1.4 |
| Channel domain | 0 | 0.01 | 0.02 | 1.4 |

*Table 5-6 Hydraulic conductivity parameters for Compartment I (Breitenau)*

| Material | Hydraulic conductivity | |
|---|---|---|
| | Saturated $K_s$ [cm/d] | Pore-connectivity $l$ [-] |
| Silt | 200 | 0.5 |
| Gravel (drain layer) | 700 | 0.5 |
| Matrix-domain | 10 | 20 |
| Channel domain | 30,000 | 0.5 |

Additionally, the anisotropy of the matrix domain concerning the hydraulic conductivity $K^A_h$ (ratio between horizontal and vertical hydraulic conductivity $K_h/K_v$) had to be set to 2.0, and the scaling factor for the pressure head $\alpha_h$ of the preferential flow path to a value of 5. These values provided good agreement between simulated and observed leachate discharges (Figure 5-7).

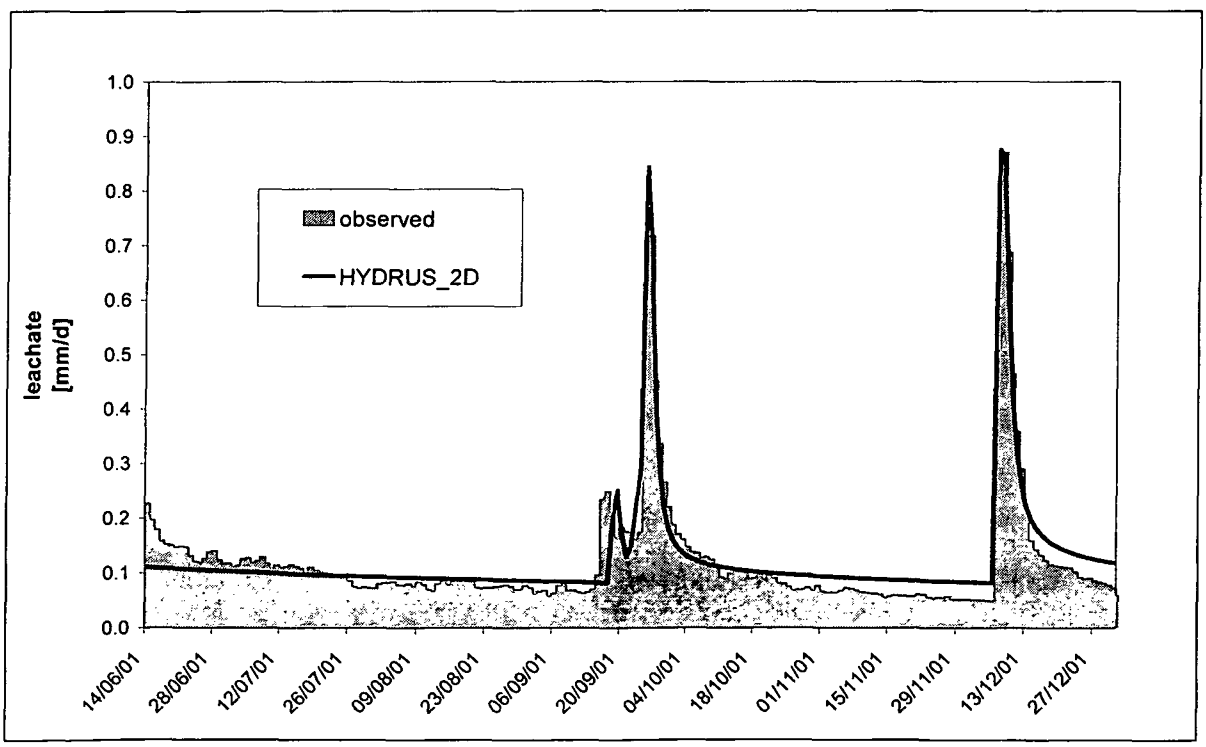



*Figure 5-7 Observed and simulated leachate discharge for the calibration period (Compartment I)*

The quality grade (according to Gaussian sum of square error) for the calibration period gives a mean discrepancy of 0.03 mm/d between observed and predicted data.

The extrapolation of the model shows that predicted and observed leachate discharges are close even beyond the calibration period (Figure 5-8). This is remarkable since maximum flow rates are nearly 10 times higher during the validation period compared to the discharge rates used for the calibration. The mean deviation of the prediction from the observed discharge is around 0.25 mm/d. Differences between model results and observation (February and May 2002) may attribute to uncertainties of the meteorological data, as measurements from nearby stations and not from the landfill site itself served as input data.

Figure 5-9 compares calculated and measured cumulative discharge during the period from June 2001 till July 2002. The model predicts a total leachate amount of 105 mm which is close to the observed value of 101 mm. The maximum error did not exceed 17 % of the observed discharge.

In general it can be postulated that the water flow model was validated successfully at Compartment I.

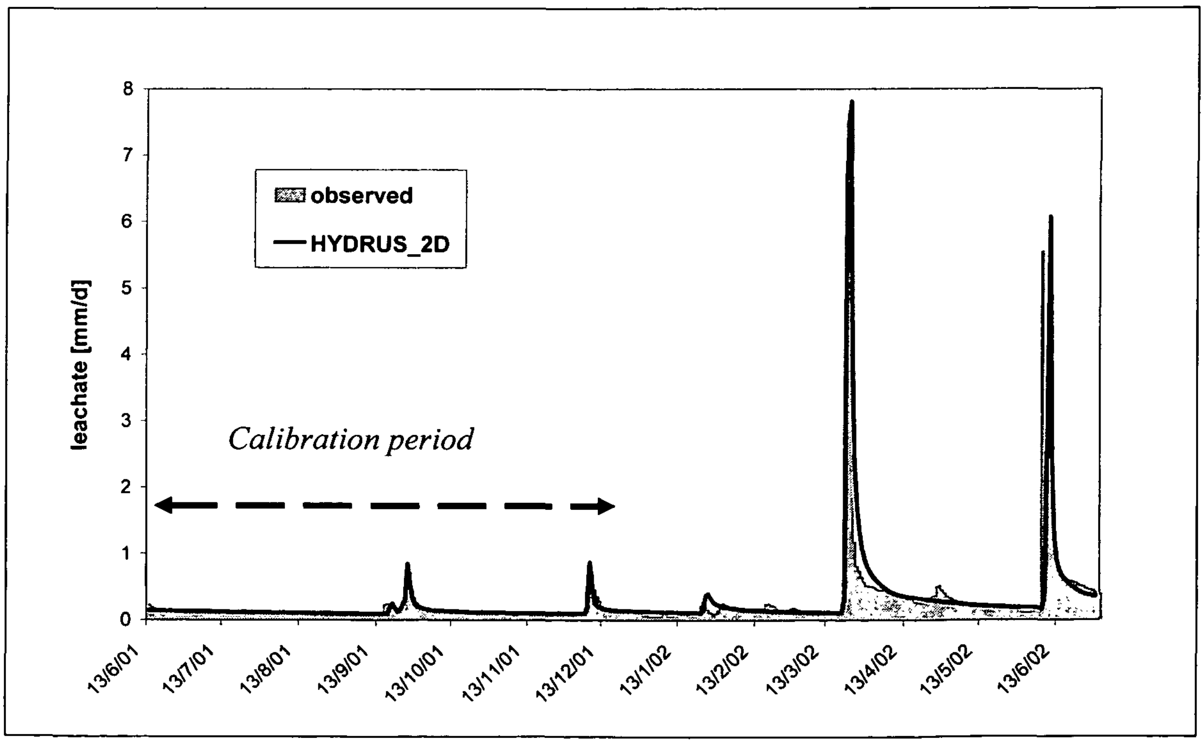



***Figure 5-8 Observed and simulated leachate discharge (Compartment I)***

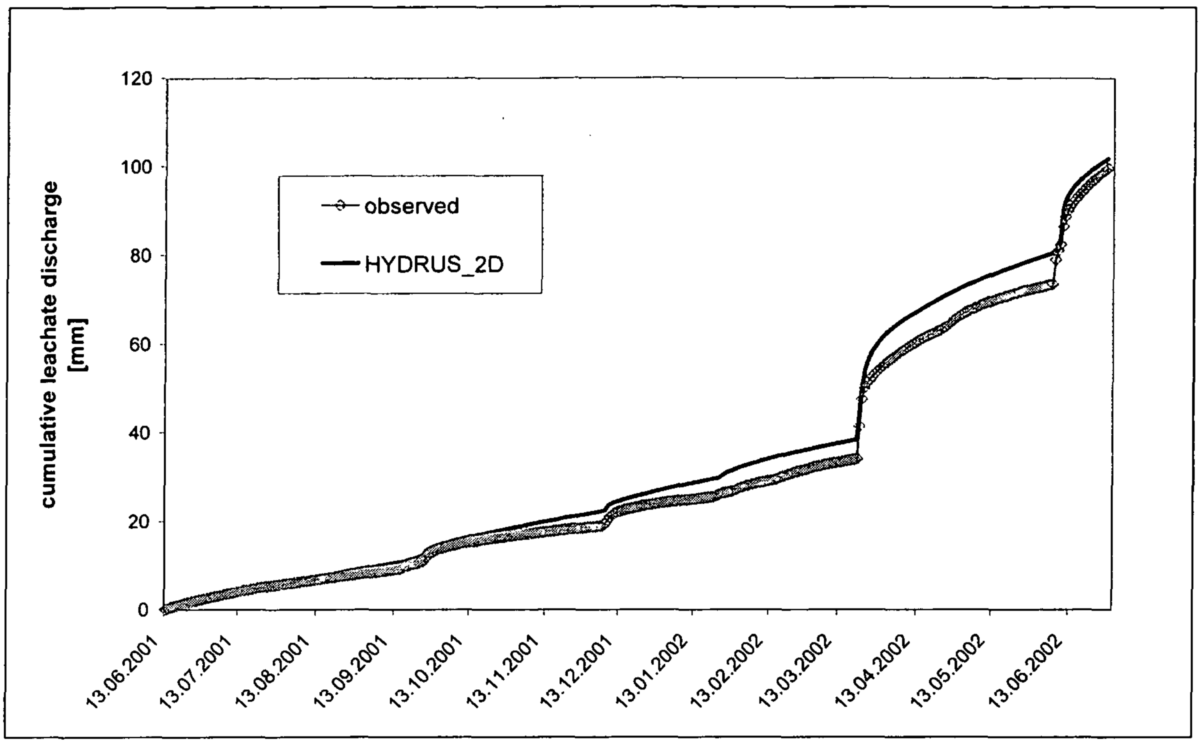



***Figure 5-9 Observed and simulated cumulative leachate discharge (Compartment I)***

### 5.1.4. Results of Compartment II

For Compartment II two separate water flow simulations had to be conducted due to the different capping systems within this compartment (see Figure 5-10). The results of the simulations were weighted according to the surface areas of the different cover layers and added up. This procedure made the calibration of the water flow model extremely difficult and time consuming, as more parameters had to be adjusted and probably several calibration optima exist. The calibration period had to be extended (compared to C I) till the end of June 2002, since no influence of precipitation on the leachate discharge was observable until spring 2002. Thus, the whole available time series was required for the calibration of the model. The lack of a further data set made it impossible to further validate the model.

The maximum root depth of the vegetation was assumed to be 15 cm and 45 cm at Compartment II/1 (gravel surface) and II/2 (compost surface), respectively.

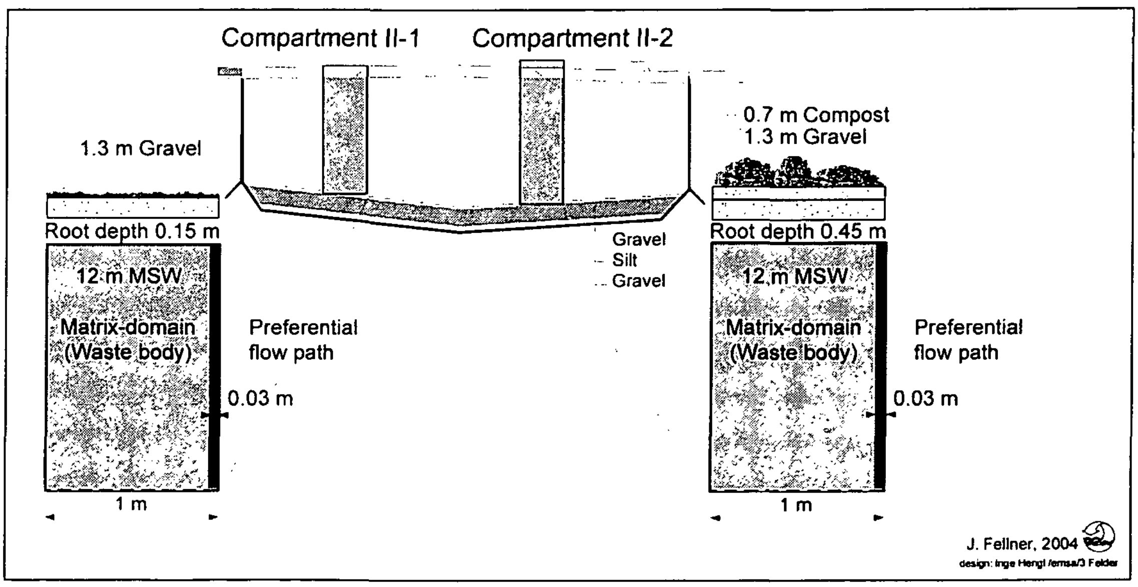



*Figure 5-10 Simplified model system (Compartment II)*

Within the scope of model calibration the following values for the hydraulic parameters of the different "materials" have been determined:

*Table 5-7* ***Water retention parameters for Compartment II (Breitenau)***

| Material | Water content | | Retention | |
|---|---|---|---|---|
| | Residual $\theta_r$ [-] | Saturated $\theta_s$ [-] | Coefficient $\alpha$ [1/cm] | Exponent $n_g$ [-] |
| Compost | 0.20 | 0.5 | 0.015 | 1.2 |
| Gravel (agglutinated) CII/1* | 0.03 | 0.29 | 0.03 | 1.5 |
| Gravel (agglutinated) CII/2* | 0.04 | 0.23 | 0.04 | 1.5 |
| Matrix domain | 0.15 | 0.5 | 0.02 | 1.4 |
| Channel domain | 0 | 0.01 | 0.02 | 1.4 |

*Table 5-8* ***Hydraulic conductivity parameters for Compartment II (Breitenau)***

| Material | Hydraulic conductivity | |
|---|---|---|
| | Saturated $K_s$ [cm/d] | Pore-connectivity l [-] |
| Compost | 300 | 0.5 |
| Gravel (agglutinated) CII/1* | 10 | 0.5 |
| Gravel (agglutinated) CII/2* | 7 | 0.5 |
| Matrix domain | 10 | 23 |
| Channel domain | 2,000 | 0.5 |

**Diverse parameter values for the gravel layer within Compartment II are explained by different degrees of agglutination. The gravel underlying compost at Compartment II/2 exhibits a higher level of agglutination and shows therefore less porosity and conductivity.*

Best match between observed and predicted leachate outflow was yielded setting the anisotropy of the hydraulic conductivity $K^A_h$ to 0.55 and the pressure head scaling factor of the channel domain to 1.9. Figure 5-11 presents measured and calculated leachate discharge versus time. The hydrographs coincide remarkably. The average deviation of the simulated water flow from the observed values was less than 16 % (according to Gaussian sum of square error). The difference refers mainly to a slight delay of the simulated discharge peaks, as the cumulative outflow values (Figure 5-11) agree well.

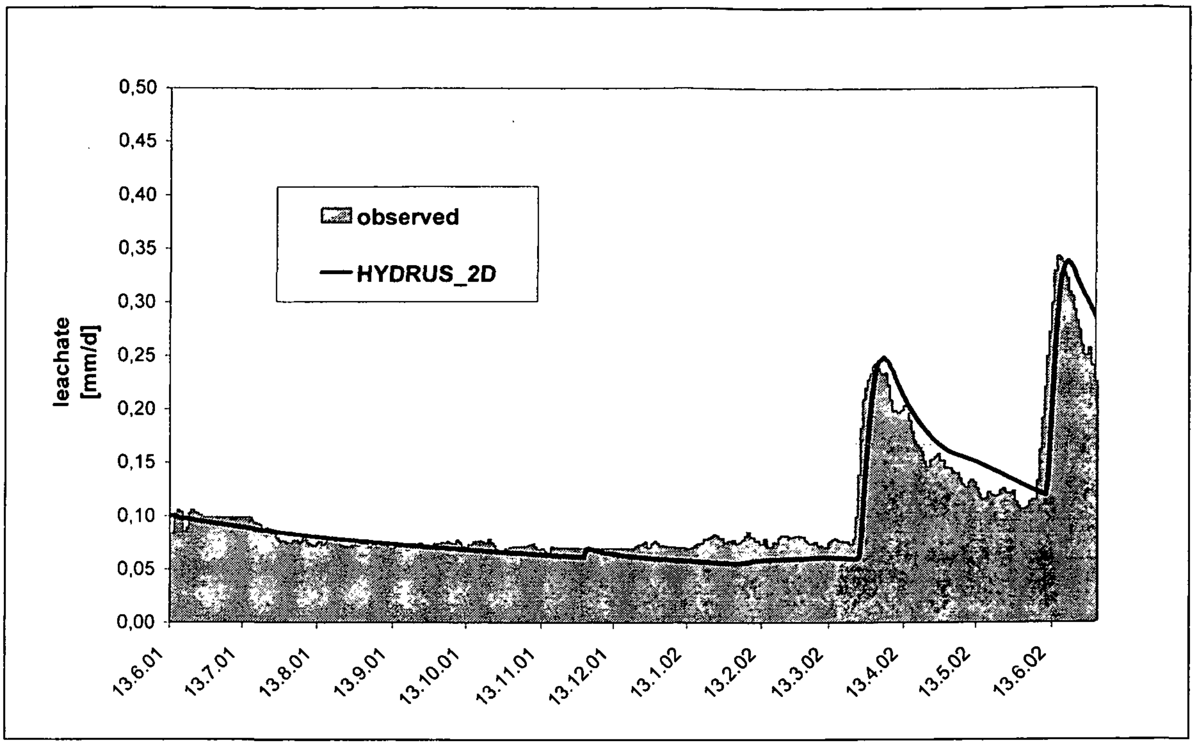



***Figure 5-11 Observed and simulated leachate discharge (Compartment II)***

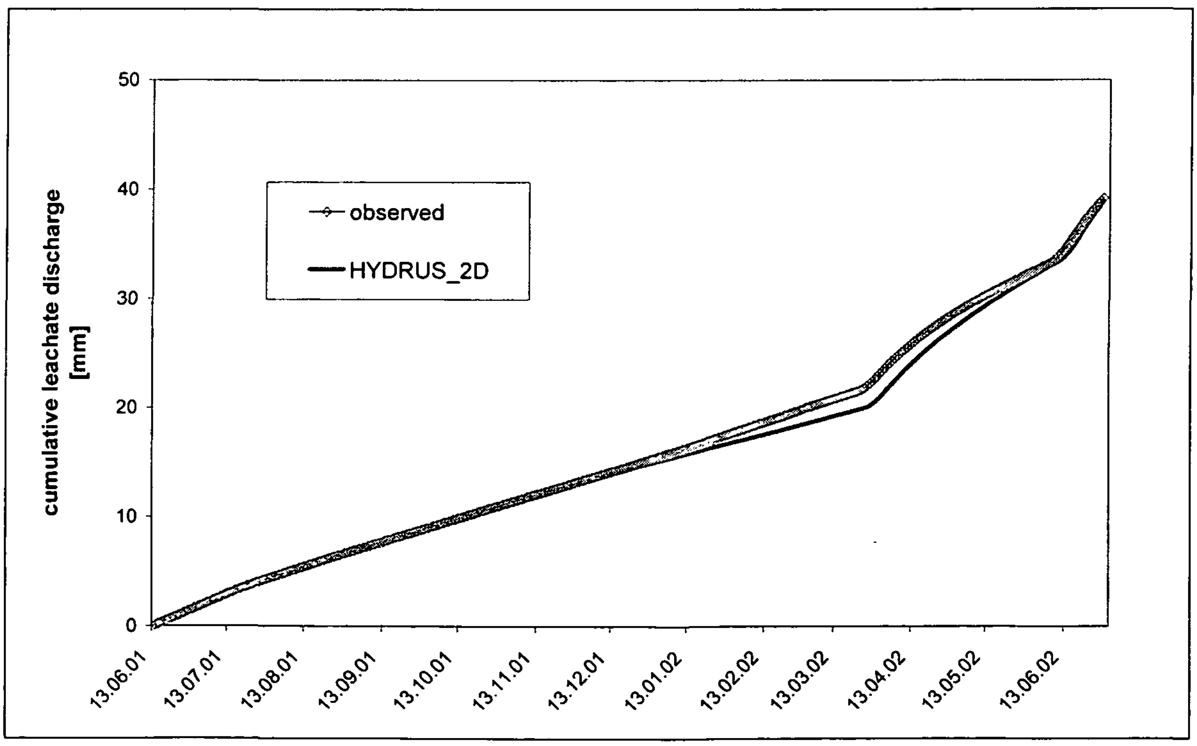



***Figure 5-12 Observed and simulated cumulative leachate discharge (Compartment II)***

### 5.1.5. Results of Compartment III

The leachate hydrograph observed at Compartment III is similar to that of Compartment II. No significant effect of rainfall events on discharge is observable for the time from June 2001 till July 2002. All precipitation evaporated due to the dry weather conditions and the capping system with a dense vegetation cover. Only close to the end of the observation period (June 2002) an increase of the leachate discharge caused by precipitation was noticed. Therefore, the recorded data set only enables to calibrate the water flow model. A following validation of the calibrated model would require a further time series that include changes in leachate generation rate. The maximum root depth representative for the vegetation cover of Compartment III was set analogous to C II/2 to 45 cm.

The calibration of the model results in the parameter values given in Table 5-9 and Table 5-10. These figures however, must be evaluated taking into account that almost no influence of rainfall on the leachate discharge was observable during the calibration period. Consequently, the performed calibration of the model is of low reliability.

***Table 5-9 Water retention parameters for Compartment III (Breitenau)***

| Material | Water content | | Retention | |
|---|---|---|---|---|
| | Residual $\theta_r$ [-] | Saturated $\theta_s$ [-] | Coefficient $\alpha$ [1/cm] | exponent $n_g$ [-] |
| Compost | 0.2 | 0.5 | 0.015 | 1.2 |
| Gravel (agglutinated) | 0.04 | 0.23 | 0.04 | 1.5 |
| Matrix-domain (waste body) | 0.15 | 0.5 | 0.02 | 1.4 |
| Preferential flow path | 0 | 0.01 | 0.02 | 1.4 |

***Table 5-10 Hydraulic conductivity parameters for Compartment III (Breitenau)***

| Material | Hydraulic conductivity | |
|---|---|---|
| | Saturated $K_s$ [cm/d] | Pore-connectivity l [-] |
| Compost | 300 | 0.5 |
| Gravel (agglutinated) | 7 | 0.5 |
| Matrix-domain (waste body) | 5 | 26 |
| Preferential flow path | 500 | 0.5 |

The ratio between horizontal and vertical conductivity (anisotropy) of the matrix domain was aligned to 0.3. Additionally, the scaling factor of the pressure head $\alpha_h$ for the channel domain was adjusted to 1.2 to achieve an agreement between simulated and observed leachate hydrographs.

Figure 5-13 shows predicted and measured discharge versus time. A good match between simulation results and observations was achieved. However, the capability of the calibrated model for predicting future leachate generation must be validated with another data set. Investigations within the scope of an ongoing research project "A New Method to Characterize the Stability of Old, Large Size landfills" (Döberl et al., 2004) will enable to evaluate the reliability of the calibrated water flow model at Compartment III.

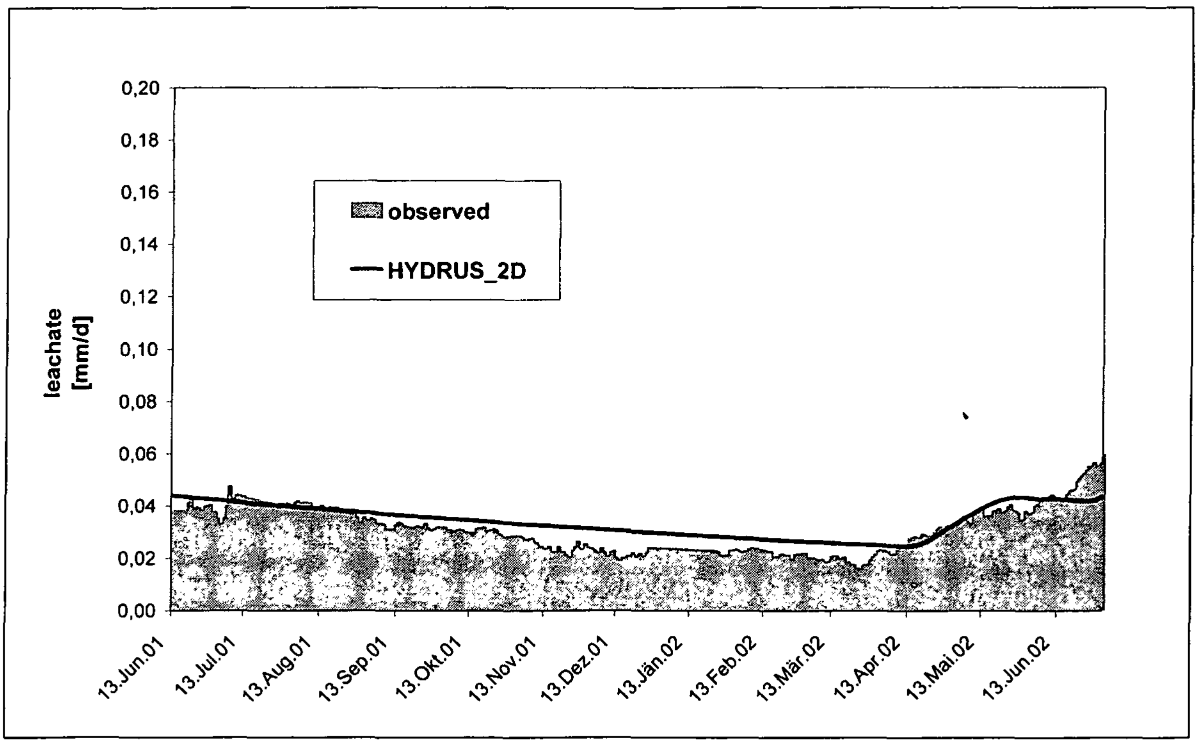



***Figure 5-13 Observed and simulated leachate discharge (Compartment III)***

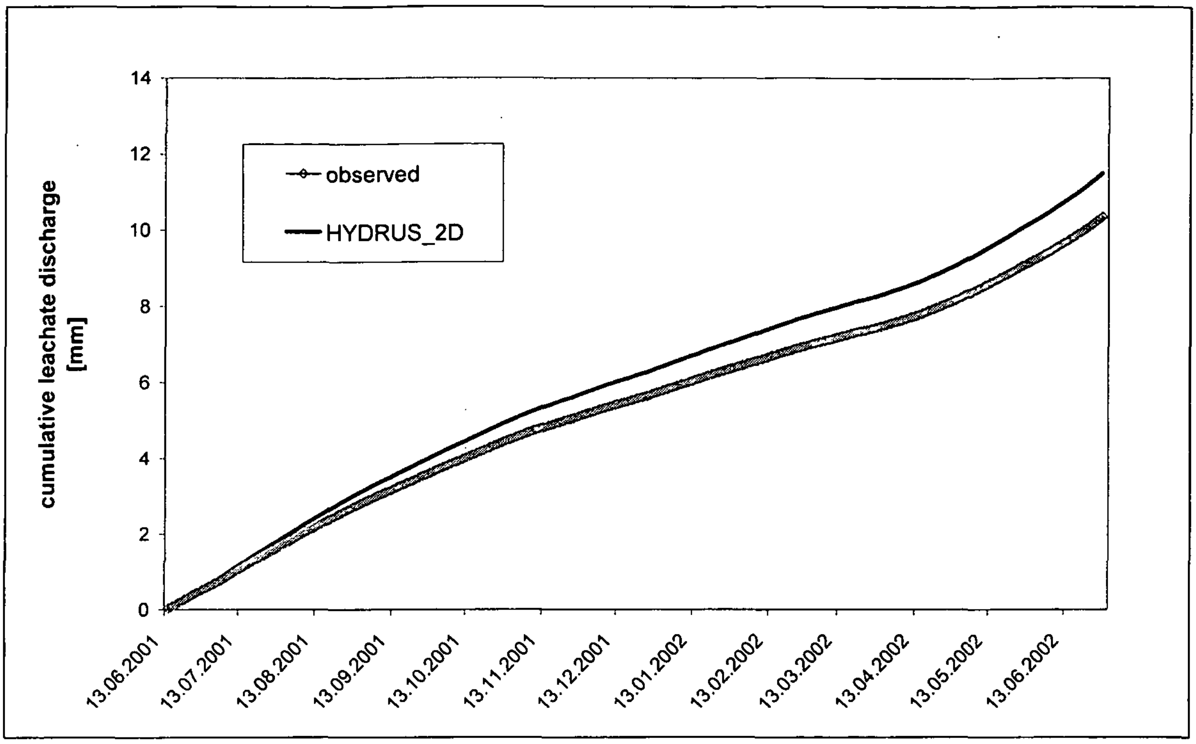



*Figure 5-14 Observed and simulated cumulative leachate discharge (Compartment III)*

## 5.2. *Case study Spillepeng test cells*

### 5.2.1. Site description

The second case study was carried out using data from MSW test cells in Sweden. The considered landfills were constructed in 1988 at the Spillepeng landfill site in the city of Malmö.

The purpose of the original project (Nilsson et al., 1991) was to evaluate the dependence of biogas production on waste composition. Altogether six test cells, each with different waste composition as shown in Table 5-11, were constructed and operated over seven years, from 1989 till 1995. The volume of each cell is approximately 8,000 m³ and the cells contain about 4,000 tons of waste. The bottom dimensions are 35×35 m and the landfill surface is sloping, so that the height is decreasing from about 10 to 2 m (Figure 5-15).

The cells have been covered immediately after landfilling with 0.5 m clay. One year after closure in August/September 1990 an additional cover of 0.5 m plant soil was placed. Grass was sown in October 1990 and the vegetation has become established in the summer of 1991 (Nilsson et al., 1992).

***Table 5-11 Waste composition and characteristics of the Spillepeng test cells (Nilsson et al., 1997)***

| | Waste composition | Volume [m³] | Mass [ton] | Average height [m] | Height (incl settlements) [m] | Wet density [kg/dm³] | Init. water content [m%] WS |
|---|---|---|---|---|---|---|---|
| **Cell 1** | Household - (70%) and industrial waste (30 %) | 7,400 | 3,400 | 6.0 | 5.7 | 0.48 | 23±5 |
| **Cell 2** | Household - (70%), industrial waste (30%) and sewage sludge (5%) | 6,800 | 3,200 | 5.6 | 5.3 | 0.53 | 26±5 |
| **Cell 3** | Household waste enriched with food waste fractions, horse manure | 7,600 | 3,500 | 6.2 | 5.9 | 0.48 | 20±5 |
| **Cell 4** | Household waste (100%) | 8,000 | 5,200 | 6.5 | 6.2 | 0.68 | 36±5 |
| **Cell 5** | Household waste (100%) and sewage sludge (5%) | 8,400 | 4,800 | 6.9 | 6.5 | 0.62 | 34±5 |
| **Cell 6** | Household waste (100%) | 7,600 | 5,200 | 6.2 | 5.9 | 0.72 | 33±5 |

The bottom liner of the cells consists of 0.5 m clay and a geomembrane liner. For the protection of the plastic liner, sand layers of 20 cm were placed below and above the geomembrane. The cross section of the cells is shown in Figure 5-15.

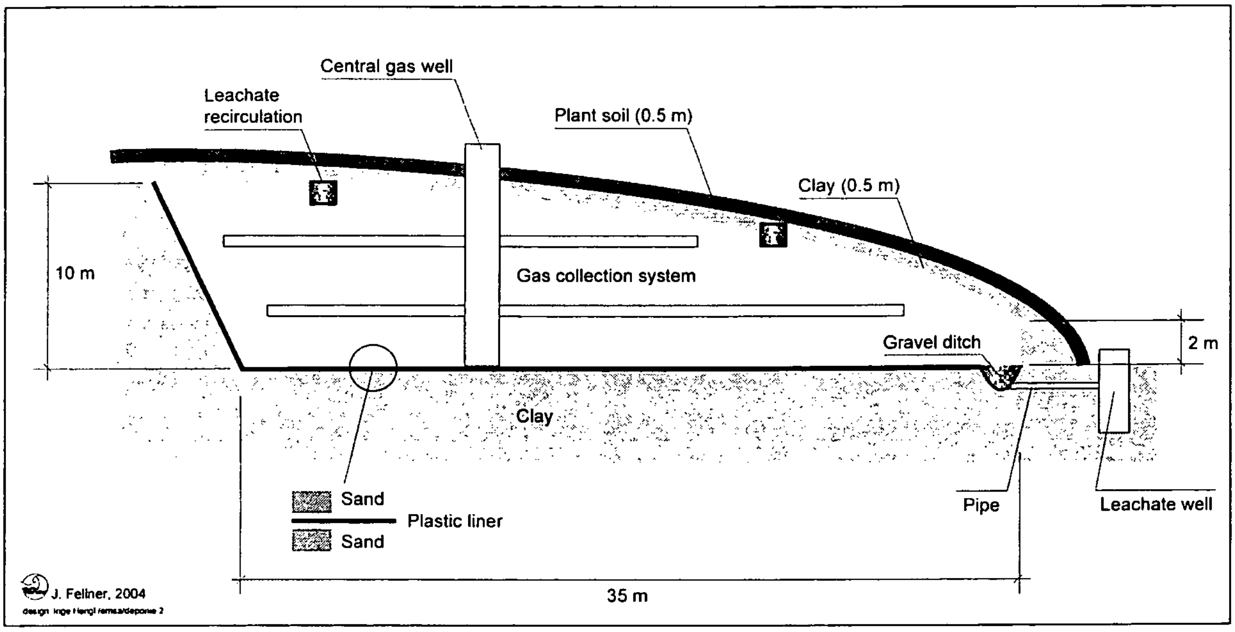



*Figure 5-15 Spillepeng test cell construction – cross section (Åkesson & Nilsson, 1997)*

Leachate is collected at the lower end of the cell, where a gravel ditch has a lined connection (PVC-pipe) to a leachate well of 2 m³. The quantity of generated leachate was determined by measuring the water level in the leachate tank. The tank was emptied manually before it became full. At high flow rates, the water level in the tank could exceed the level of the pipe connecting the well to the gravel ditch inside the test cell, thereby preventing the leachate from draining. Consequently, additional discharge was draining when the well was emptied. These extra volumes were quantified through the time of pumping. In the case that the well was not emptied regularly a larger amount of leachate was retained inside the test cells (within the gravel ditch) and it was impossible to get information on the temporal variation in the collected leachate. To avoid retention of water inside the cells (in particular inside the gravel ditch) during periods with high rate of leachate generation, the plant was upgraded in December 1994, so that the wells were emptied automatically. Since this date the recorded pumping time was used to determine the leachate discharge.

For the observation period from January 1989 to December 1995 the recorded data of the leachate amount was available on a weekly basis. Some periods however had a lower temporal resolution. In particular measurements were missing for the time between July 1990 and January 1991. Furthermore, the emptying of leachate wells was disregarded in the period from February till August 1994. Well documented data with high temporal resolution exist for the last year of observation (1995), after the upgrading of the leachate management system.

### 5.2.2. Input information

#### 5.2.2.1. Meteorological data (Spillepeng)

The meteorological data such as precipitation, temperature, humidity, wind speed and sunshine hours (necessary to calculate the potential evapotranspiration), were obtained from the nearby meteorological stations Malmö (3 kilometres distance) and Lund (around 15 kilometres). In particular values for precipitation, temperature and humidity were taken from Malmö, whereas information on sunshine hours was only available from the station in Lund. During periods with malfunctioning of the meteorological station in Malmö, the data from Lund had to be consulted.

The reference evapotranspiration $ET_0$ for the Spillepeng site was estimated using the method after Penman-Moneith (Bevan, 1979). In order to obtain the potential evapotranspiration $ET_p$ of the considered vegetation the reference values $ET_0$ were multiplied by a constant crop coefficient $K_c$ of 1.15 (dense vegetation cover) for the growing period (April till October). Beyond this period a dependence of $K_c$ from the reference evapotranspiration $ET_0$ according to Allen et al. (1998) was used (see Figure 5-4).

Surface runoff from the landfill cover was only measured for short time (from October 1993 till March 1994) and assumed to be negligible (Bendz et al., 1997). However, the distinct slope (15 %) of the landfill surface calls for considering overland flow. In particular short time after landfill closure (no vegetation cover) a considerable amount of runoff was observed by the operational staff of the Spillepeng site (Andersson, 2003). As measured data were not available, runoff was calculated according to the common SCS-curve number method (Soil Conservation Service, 1973). This approach accounts for the vegetation cover, the water content of the cover layers and the slope of the surface.
The results of the calculations show that with increasing time after landfill closure and therefore denser crop cover runoff is declining (Figure 5-16).

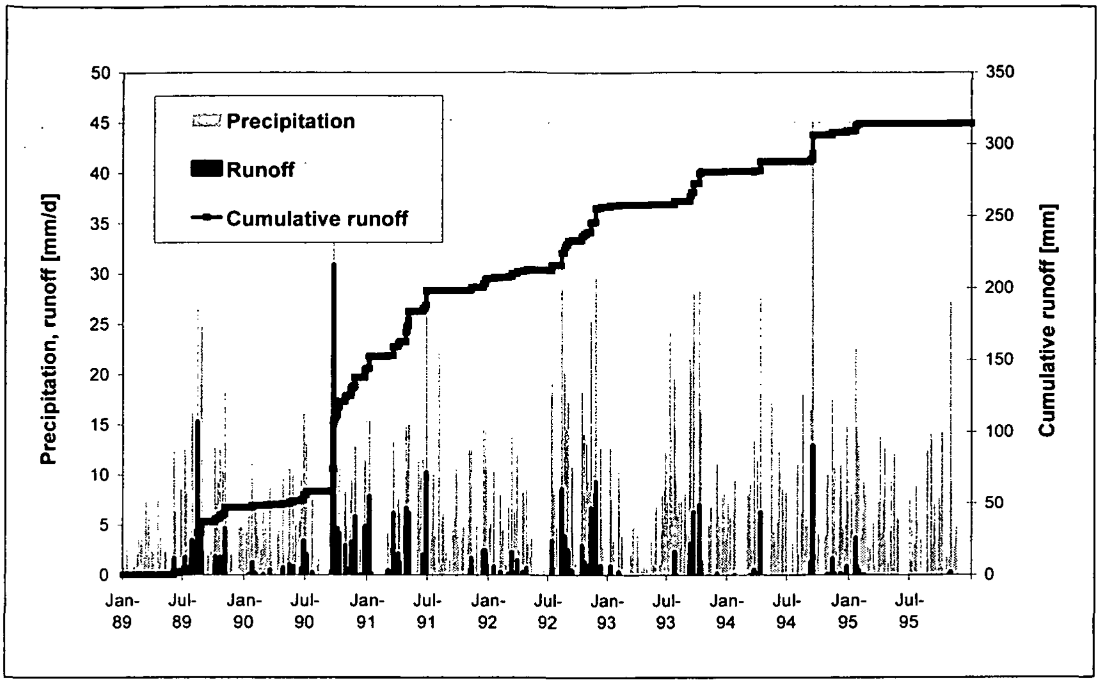



***Figure 5-16 Precipitation and estimated runoff (Spillepeng test cells)***

### 5.2.2.2. Hydraulic properties

The required hydraulic characteristics of the cover materials were determined by field and laboratory experiments performed by the author during a research visit at Lund University (lasting from January till June 2003). The hydraulic conductivity was measured using infiltration tests. So-called inversed bore-hole tests (Klute, 1986) were performed. The porosity, the bulk density and the retention characteristics (curve) were determined in the laboratory using undisturbed soil samples. The pressure cell method after Richards (1941) was applied to derive the relation (retention characteristics) between matrix potential (pressure head) and water content of the two cover soils. In Table 5-12 and Table 5-13 the results of the hydraulic investigations are presented.

***Table 5-12 Conductivity, porosity and density of the landfill cover layers (Spillepeng cells)***

| Cover material | Saturated hydraulic conductivity $K_s$ [cm/d] | Porosity n [-] | Bulk density $\rho_d$ [kg/dm³] |
|---|---|---|---|
| „Plant soil“ | 20 – 120 | 0.33 – 0.41 | 1.59 – 1.75 |
| „Clay“ | 0.6 – 8 | 0.34 – 0.43 | 1.52 – 1.80 |

*Table 5-13 Estimated van Genuchten parameters of the landfill cover layers (Spillepeng cells)*

| Cover material | Methods | van Genuchten parameter | | | |
|---|---|---|---|---|---|
| | | Residual water content $\theta_r$ [-] | Saturated water content $\theta_s$ [-] | Parameter $\alpha$ [1/cm] | Parameter $n_g$ [-] |
| „Plant soil" | Pressure cell, curve fitting | 0.04 – 0.09 | 0.29 – 0.35 | 0.042 – 0.081 | 1.44 – 1.66 |
| „Clay" | Pressure cell, curve fitting | 0.07 – 0.15 | 0.34 – 0.39 | 0.010 – 0.023 | 1.25 – 1.39 |

First estimates concerning the hydraulic characteristics of the waste "domains" are corresponding to those for the landfill Breitenau. Also the same calibration procedure (trial and error method with graphical evaluation) was used (e.g. see chapter 5.1.3). Due to the minor leachate discharge during the first years and the poor temporal resolution of available data during this time, the calibration was performed using the time series from September 1994 till December 1995. This period coincides with the phase of automatic emptying of the leachate collection wells. Additionally the total cumulative amount of generated leachate during the whole observation period (1989-1995) was used for calibration purposes.
The introduced water flow model was calibrated at Cell 1, Cell 2 and Cell 4 only. For the other cells either reasonable suspicions that the leachate collection system was malfunctioning exist (Cell 3 and Cell 5, Bendz et al., 1997) or the documentation about operational data was insufficient (leachate recirculation at Cell 6).

### 5.2.3. Results of Cell 1

Analogous to the water flow modeling at the Breitenau landfill, separate simulations were carried out for the landfill cover and the waste body. The results of the simulations for the landfill cover (seepage output of HYDRUS-2D) served as input information for the hydrologic system waste body that consists of the matrix domain and the channel domain. The calculations for this system were performed for an average landfill profile (height for Cell 1 equals 5.7 m) with a width of 0.5 m (see Figure 5-17). The reduction of the profile width compared to the simulations for the landfill Breitenau is necessary to get parameter values that are comparable to those obtained for landfills of bigger height. The root depth of the vegetation was set according to field investigations to 30 cm, whereby in order to simplify matters a constant distribution of roots over the depth was assumed. The initial mass water content of the landfilled waste in Cell 1 was reportedly around 23 % (referred to wet mass). This results for a given waste density of 0.48 kg/dm³ in a volumetric water content θ of 0.11. This value was used as initial condition for the water flow simulations.

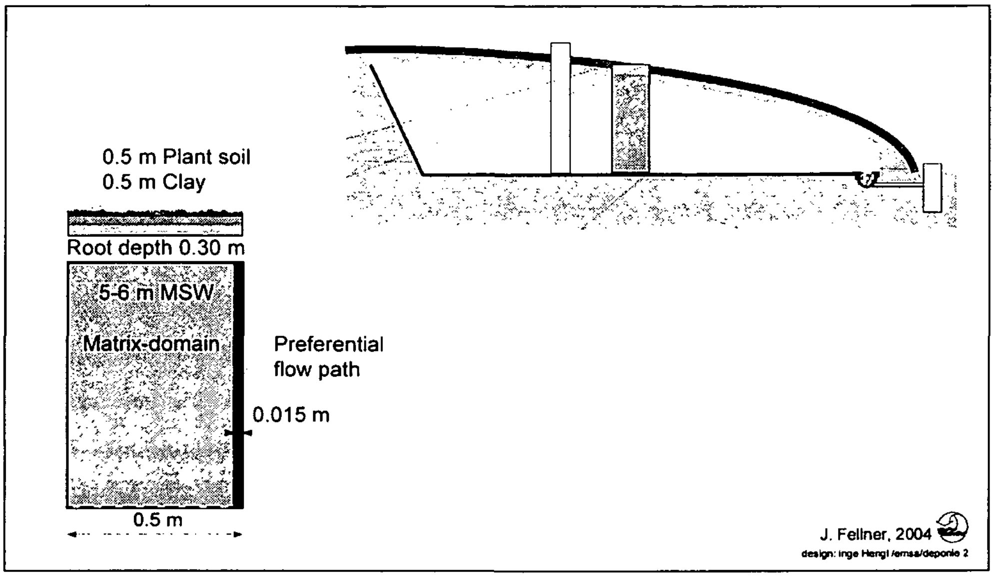



*Figure 5-17 Simplified model system (Spillepeng test cells)*

Best match between predicted and observed leachate discharge was achieved using the parameter values of the landfill cover layers and the waste domains given in Table 5-14 and Table 5-15.

*Table 5-14 Water retention parameters for Cell 1 (Spillepeng landfill)*

| Material | Water content | | Retention | |
|---|---|---|---|---|
| | Residual $\theta_r$ [-] | saturated $\theta_s$ [-] | Coefficient $\alpha$ [1/cm] | Exponent $n_g$ [-] |
| Plant soil | 0.06 | 0.31 | 0.05 | 1.53 |
| Clay | 0.1 | 0.35 | 0.014 | 1.26 |
| Matrix-domain (waste body) | 0.02 | 0.42 | 0.02 | 1.4 |
| Channel domain | 0 | 0.01 | 0.02 | 1.4 |

*Table 5-15 Hydraulic conductivity parameters for Cell 1 (Spillepeng landfill)*

| Material | Hydraulic conductivity | |
|---|---|---|
| | Saturated $K_s$ [cm/d] | Pore-connectivity l [-] |
| Plant soil | 50 | 0.5 |
| Clay | 5 | 0.5 |
| Matrix-domain (waste body) | 3 | 11 |
| Channel domain | 300 | 0.5 |

Additionally to the above listed parameter values, the anisotropy of the matrix domain concerning the hydraulic conductivity $K^A_h$ (representing the ratio between horizontal and vertical hydraulic conductivity $K_h/K_v$) was set to 0.45 and the scaling factor for the pressure head $\alpha_h$ of the preferential flow path to a value of 2 for best agreement between simulated and measured discharge during the calibration period (see Figure 5-18).

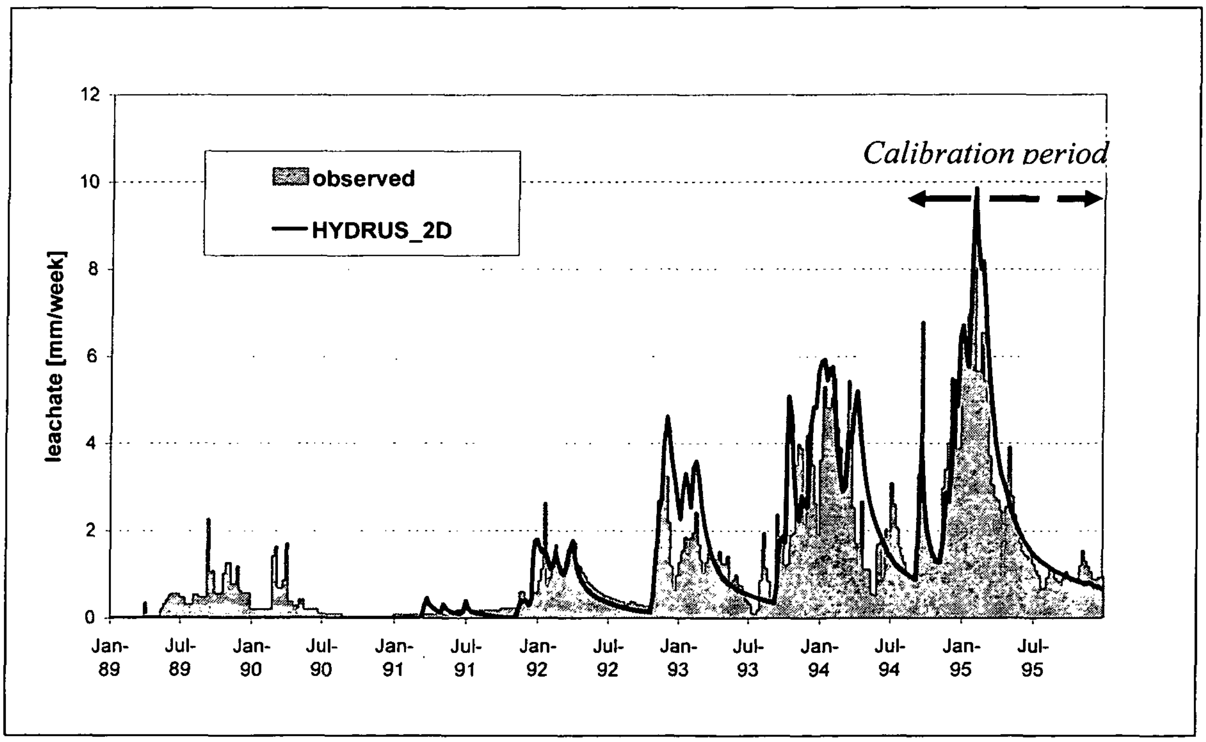



*Figure 5-18 Observed and simulated leachate discharge (Cell 1)*

The validation of the model shows that also beyond the calibration time, the discharge was predicted quite accurately. Only during the first years after landfilling (1989-1990) no leachate outflow was simulated, which is contrary to the observation. This fact is attributed to the simplification of the test cell with different heights to an average profile with a constant depth of 5.7 m, thereby neglecting areas within the cell of 2 m depth only. Water storage capacity within this lower end of the test cell is exceeded faster than the capacity of the modeled (simplified) landfill profile would admit. Thus, first discharge from the test cell occurs earlier compared to a landfill of constant depth (as assumed for the modeling effort). Furthermore, differences between measured and predicted leachate generation can be attributed to uncertainties associated with the evaluation of runoff and evapotranspiration. Some deviations (e.g. spring till summer 1994) however, refer to discontinuous emptying of the leachate collection tank, and thus, misleading observed leachate discharge. Partly misrepresented observation data is also the reason for renouncing the determination of a quality grade which evaluates the match between simulated and measured leachate generation rate.

Observed and predicted cumulative discharge (Figure 5-19) show small differences, that attribute as mentioned above on the one hand to simplifications of the landfill geometry and on the other hand to uncertainties associated with the water input into the landfill. Water storage processes within the landfill body seem to be described adequately as calculated and observed water content at the end of the simulation period are corresponding (Figure 5-24) well.

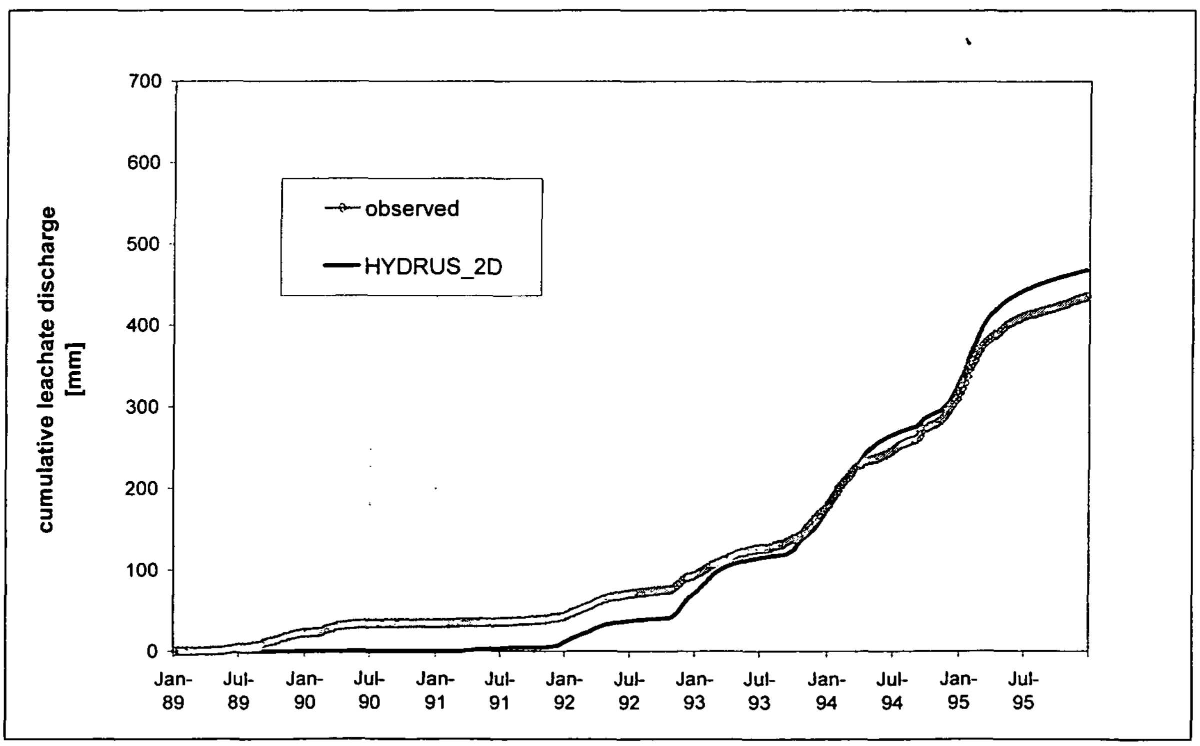



***Figure 5-19 Observed and simulated cumulative leachate discharge (Cell 1)***

### 5.2.4. Results of Cell 2

The leachate hydrograph at Cell 2 shows a similar shape to those at Cell 1. This is due to the same composition of the test cells (size, cover layers). The rate and cumulative amount of water drained from Cell 2 however, is higher compared to the discharge of Cell 1. This may be due to the different initial water content (Table 5-11), and thus, different available water storage capacities. The fact that the average landfill height of both test cells is somewhat different may also affect the water storage and the leachate generation. According to reported data the simulations were carried out with an initial volumetric water content $\theta$ of the waste matrix of 0.14.

The calibration of the water flow model was limited to hydraulic characteristics of the waste domains only, as parameters of the cover layers had already been determined for Cell 1. The results of the calibration are presented in Table 5-16 and Table 5-17.

*Table 5-16 Water retention parameters for Cell 2 (Spillepeng landfill)*

| Material | Water content | | Retention | |
|---|---|---|---|---|
| | Residual $\theta_r$ [-] | Saturated $\theta_s$ [-] | Coefficient $\alpha$ [1/cm] | Exponent $n_g$ [-] |
| Matrix-domain (waste body) | 0.02 | 0.42 | 0.02 | 1.4 |
| Channel domain | 0 | 0.01 | 0.02 | 1.4 |

*Table 5-17 Hydraulic conductivity parameters for Cell 2 (Spillepeng landfill)*

| Material | Hydraulic conductivity | |
|---|---|---|
| | Saturated $K_s$ [cm/d] | Pore-connectivity $l$ [-] |
| Matrix-domain (waste body) | 3 | 9 |
| Channel domain | 600 | 0.5 |

Additionally the calibration of the water flow model leads for the matrix domain to an anisotropy of the hydraulic conductivity $K^A_h$ of 0.6, and for the channel domain to a pressure head scaling factor $\alpha_h$ of 3.

The simulation results (Figure 5-20) indicate that the model predicts leachate generation quite accurately even beyond the calibration period. Analogously to simulations for Cell 1, deviations are noticeable only short time after landfilling as well as during periods when leachate collection wells were emptied erratically. The largest errors between measured and predicted cumulative discharge (Figure 5-21) occurred during the second year after landfill closure, when nearly no drainage was predicted by the model. During the remaining time simulated, the maximum and average errors were 30 % and 8 %, respectively.

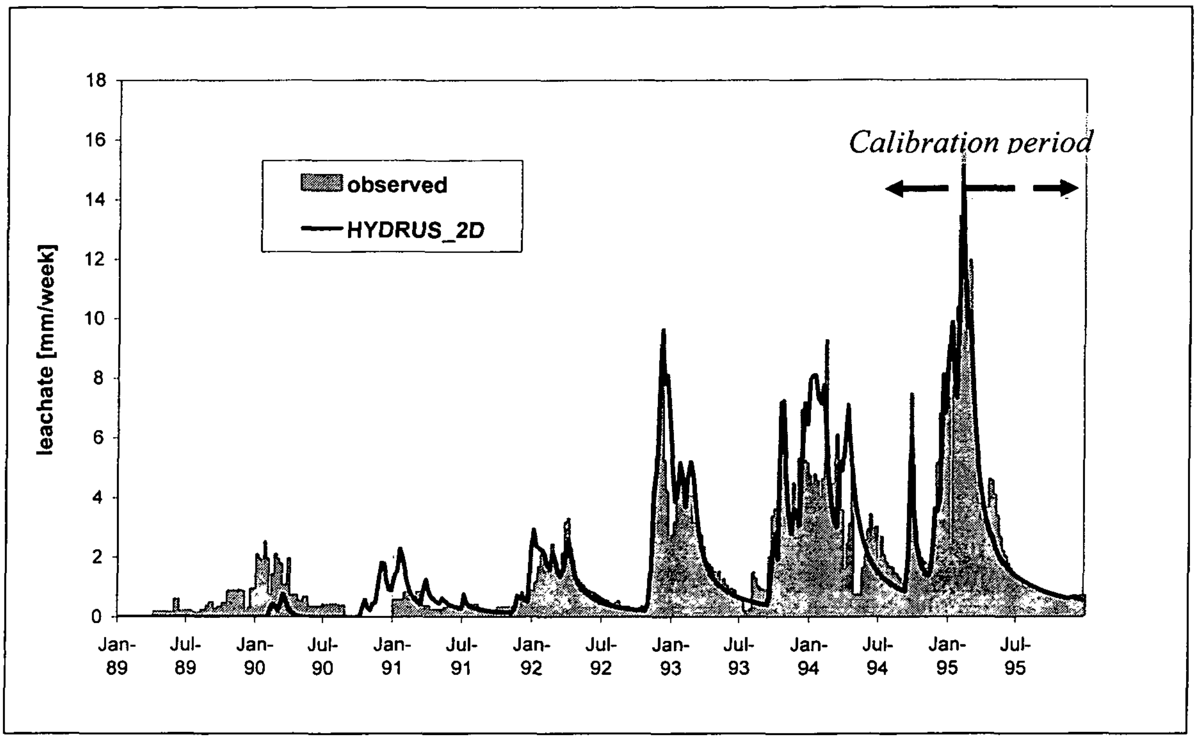



***Figure 5-20 Observed and simulated leachate discharge (Cell 2)***

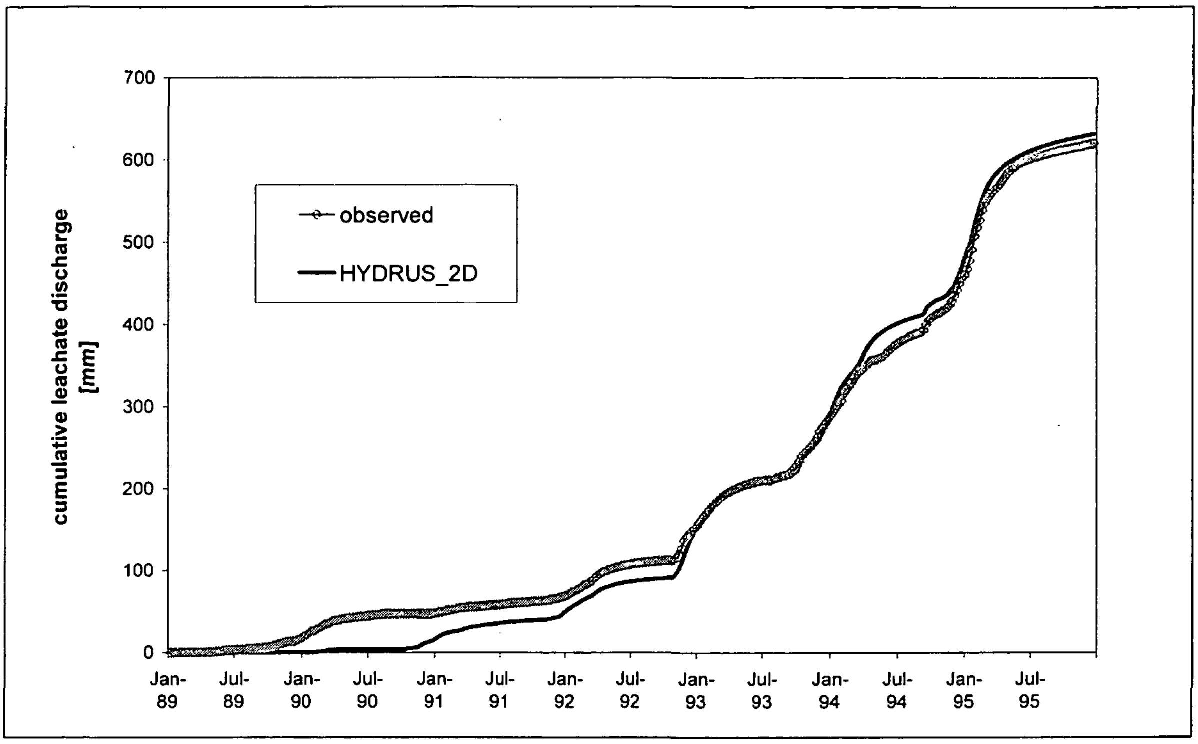



*Figure 5-21* ***Observed and simulated cumulative leachate discharge (Cell 2)***

### 5.2.5. Results of Cell 4

The average landfill height of Cell 4 is around 6.2 m. The landfilled waste shows an initial mass water content of 36 % (referred to wet mass), that corresponds to a volumetric water content of 0.25. Compared to Cell 1 and 2 the water content of the waste is higher which probably attributes to different composition (Nilsson et al., 1997). The self-evident assumption that soggier waste will generate more leachate was not confirmed by the observation. Moreover Cell 4 shows least leachate generation during the period from 1989 till 1995 (Cell 1 ~ 430 mm, Cell 2 ~ 615 mm, Cell 4 ~ 340 mm). This may be due to a higher water sorption capability of the waste landfilled or because of a more uniform water distribution within Cell 4. Both facts are associated with a larger effective water storage capacity of the landfill. The calibrated parameter values of the water flow model can provide an indication of the predominating process responsible for enhanced water retention within Cell 4 (Figure 5-24).

In order to reach best match between predicted and observed leachate discharge, model parameters had to be adjusted to the values given in Table 5-18 and Table 5-19. Furthermore, anisotropy of the hydraulic conductivity $K^A{}_h$ had to be set to 0.2 and the pressure head scaling factor $\alpha_h$ for the channel domain to 1.8.

*Table 5-18 Water retention parameters for Cell 4 (Spillepeng landfill)*

| Material | Water content | | Retention | |
|---|---|---|---|---|
| | Residual $\theta_r$ [-] | Saturated $\theta_s$ [-] | Coefficient $\alpha$ [1/cm] | Exponent $n_g$ [-] |
| Matrix-domain (waste body) | 0.1 | 0.5 | 0.02 | 1.4 |
| Channel domain | 0 | 0.01 | 0.02 | 1.4 |

*Table 5-19 Hydraulic conductivity parameters for Cell 4 (Spillepeng landfill)*

| Material | Hydraulic conductivity | |
|---|---|---|
| | Saturated $K_s$ [cm/d] | Pore-connectivity l [-] |
| Matrix-domain (waste body) | 3 | 12 |
| Channel domain | 200 | 0.5 |

Figure 5-22 represents predicted and measured leachate generation as a function of time. Predicted and observed values are close excluding the first time after landfill closure. The same phenomenon was noticed for Cell 1 and 2, and is attributed to the simplification of the landfill geometry.

Comparing the calibrated model parameters of Cell 4 with those of Cell 1 and 2 indicates that the water sorption capability, represented somehow by the retention characteristics of the matrix domain ($\theta_s$, $\theta_r$, $\alpha$, n), was enhanced. The uniformity of the flow regime, expressed by smaller values of the hydraulic anisotropy $K^A{}_h$, the scaling factor for the pressure head $\alpha_h$ and the saturated hydraulic conductivity $K_s$ of the channel domain was also enhanced. Therefore higher water storage within Cell 4 can be ascribed to both reasons: higher water sorption capability of the waste mass itself and more uniform water flow.

Predicted and measured cumulative leachate discharge parallel each other closely (Figure 5-23). Differences result only from the first years after landfilling, when the model underpredicts leachate generation. The reason therefore is once again the simplification of the landfill geometry to a rectangular profile.

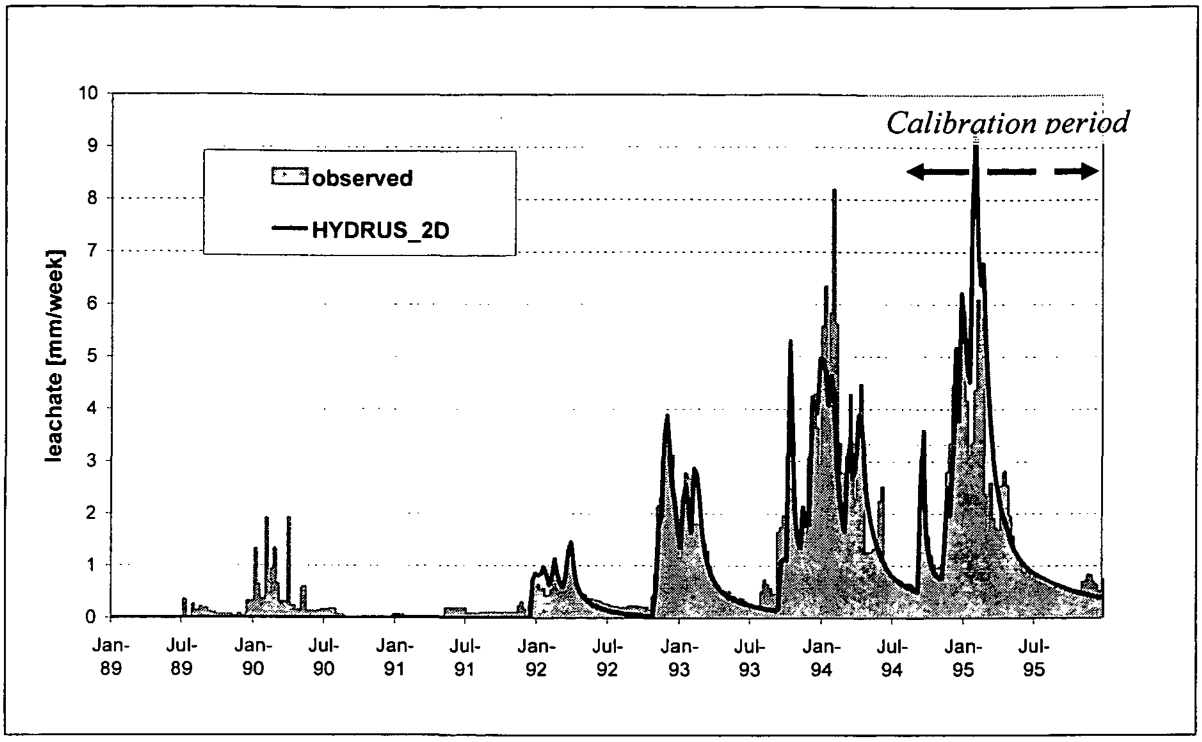



*Figure 5-22* ***Observed and simulated leachate discharge (Cell 4)***

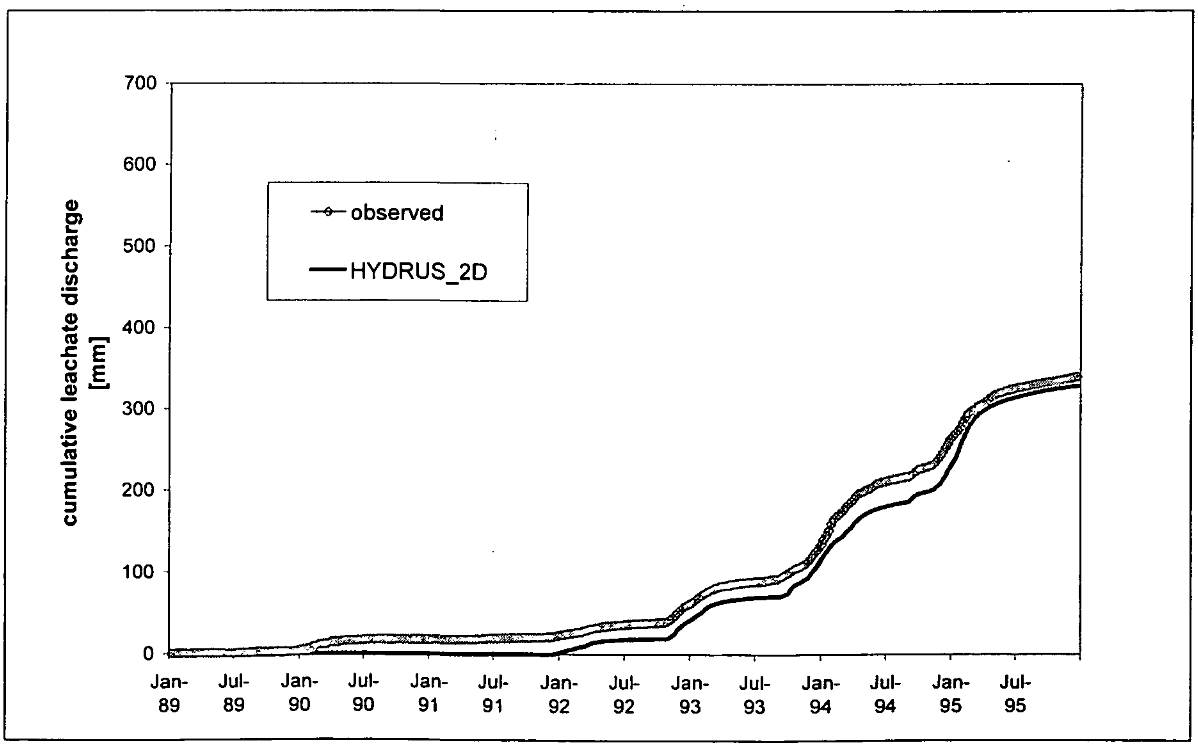



*Figure 5-23* ***Observed and simulated cumulative leachate discharge (Cell 4)***

Figure 5-24 shows predicted and observed average water content of all three cells at the end of the simulation period (December 1995). The results indicate that the model also accurately reproduces water storage processes within the landfill.

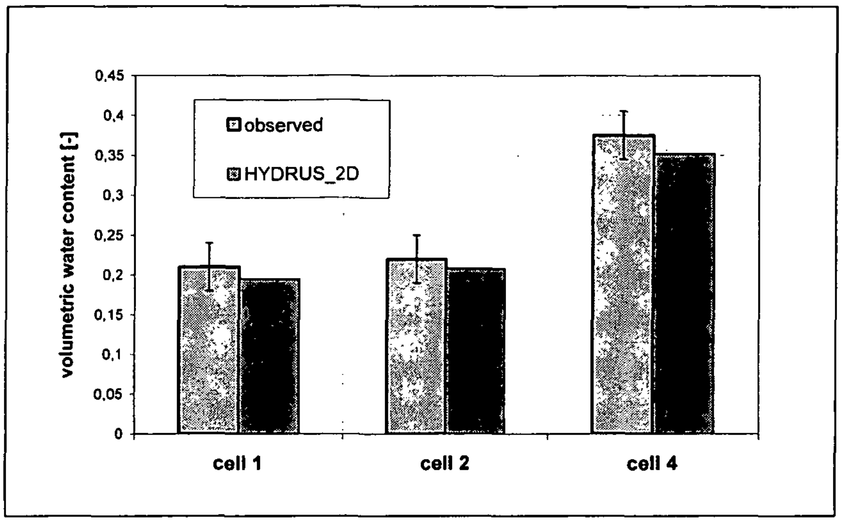



*Figure 5-24 Observed (Nilsson et al., 1997) and simulated average water content (Dec. 1995)*

## 5.3. *Sensitivity analysis*

A set of simulation runs was performed to assess the model's sensitivity to its hydraulic parameters. The sensitivity analyses focused on parameters of the waste mass (matrix and channel domain). Hydraulic parameters of the cover layers and meteorological input data have not been investigated. Within the scope of the sensitivity analyses the parameter values obtained from the calibration study of the Spillepeng test Cell 2 were taken as standard values. Each of the parameters was varied within feasible ranges (see Table 4-5), while all the other parameters were kept constant.

Changes in the shape of the discharge hydrographs (e.g. ratio between base flow and discharge peaks) as well as in the cumulative leachate discharge due to parameter variation have been investigated. Some results are presented in the following figures, whereby only a period of one year (August 1994 till July 1995) is shown in order to facilitate the differentiation of the curves. The black thick line in the following figures represents the results for the calibrated parameter set.

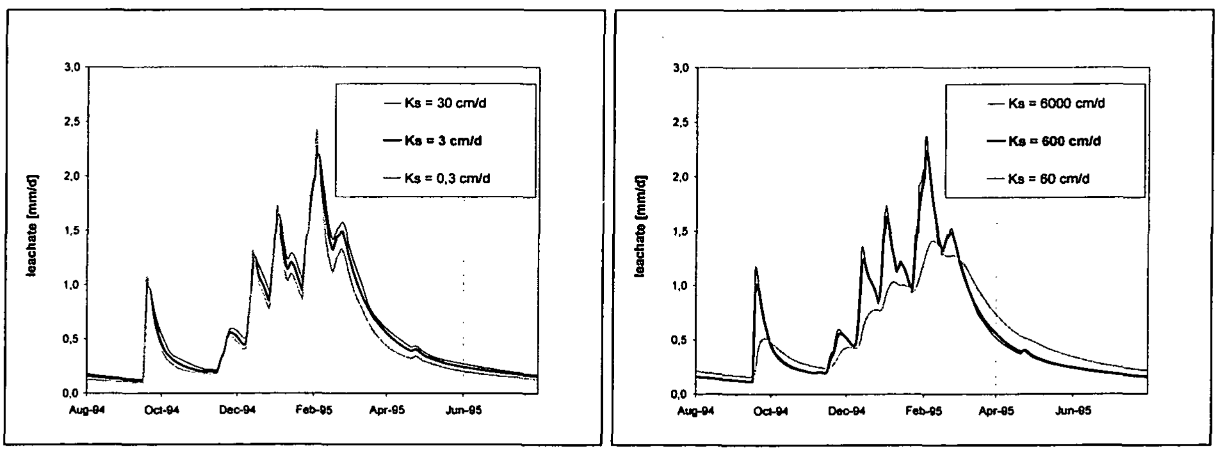



***Figure 5-25*** ***Sensitivity analysis for the saturated hydraulic conductivity $K_s$ of the matrix domain (left side) and the channel domain (right side) for Cell 2***

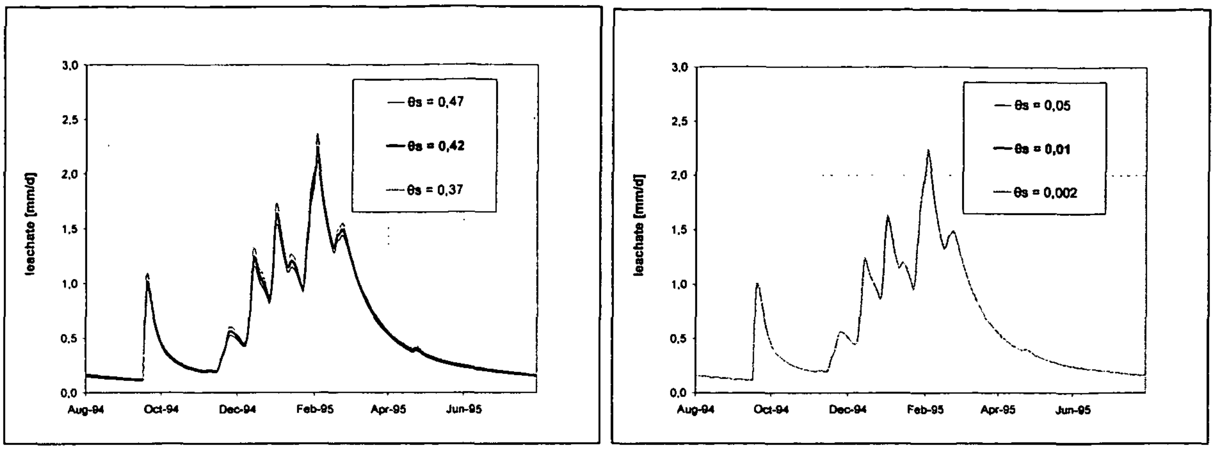



*Figure 5-26* ***Sensitivity analysis for the saturated water content $\theta_s$ of the matrix domain (left side) and the channel domain (right side) for Cell 2***

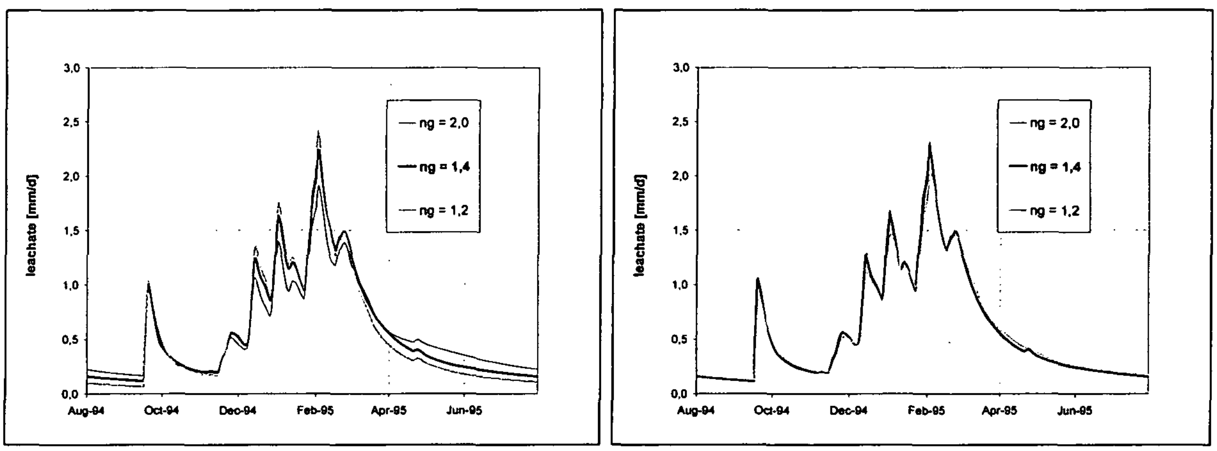



*Figure 5-27* ***Sensitivity analysis for the form parameter $n_g$ of the matrix domain (left side) and the channel domain (right side) for Cell 2***

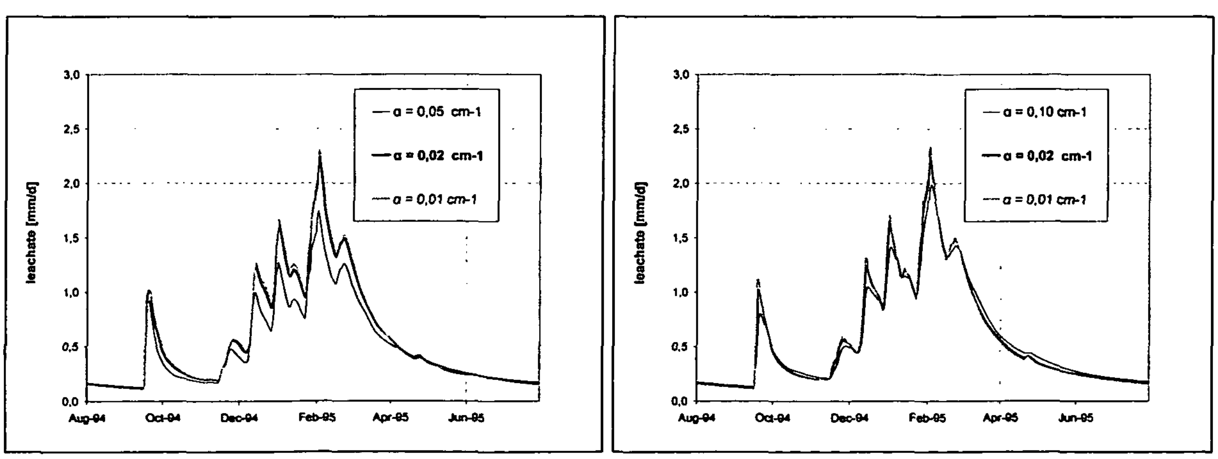



*Figure 5-28* ***Sensitivity analysis for the form parameter $\alpha$ of the matrix domain (left side) and the channel domain (right side) for Cell 2***

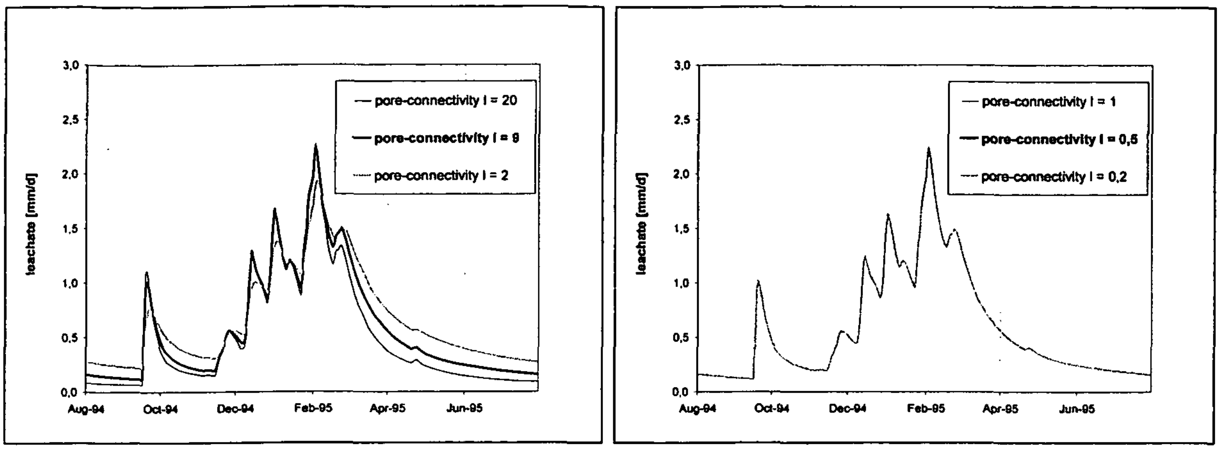



*Figure 5-29* ***Sensitivity analysis for the pore-connectivity parameter l of the matrix domain (left side) and the channel domain (right side) for Cell 2***

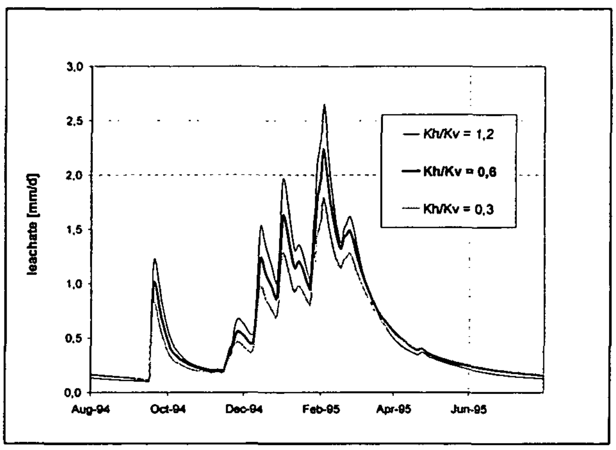



*Figure 5-30* ***Sensitivity analysis for the parameter hydraulic anisotropy $K_h/K_v$ of the matrix domain for Cell 2***

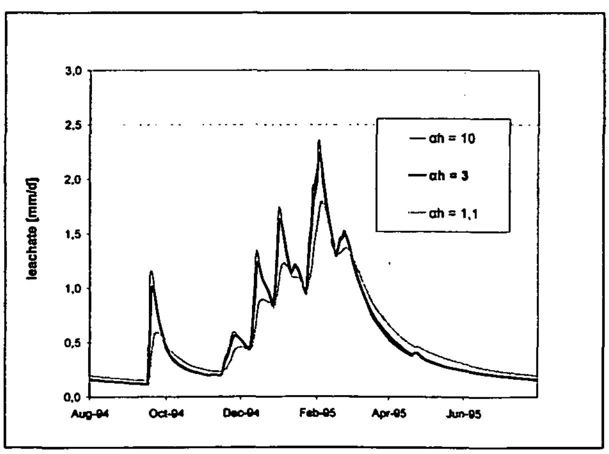



*Figure 5-31* ***Sensitivity analysis for the parameter pressure head scaling factor $\alpha_h$ of the channel domain for Cell 2***

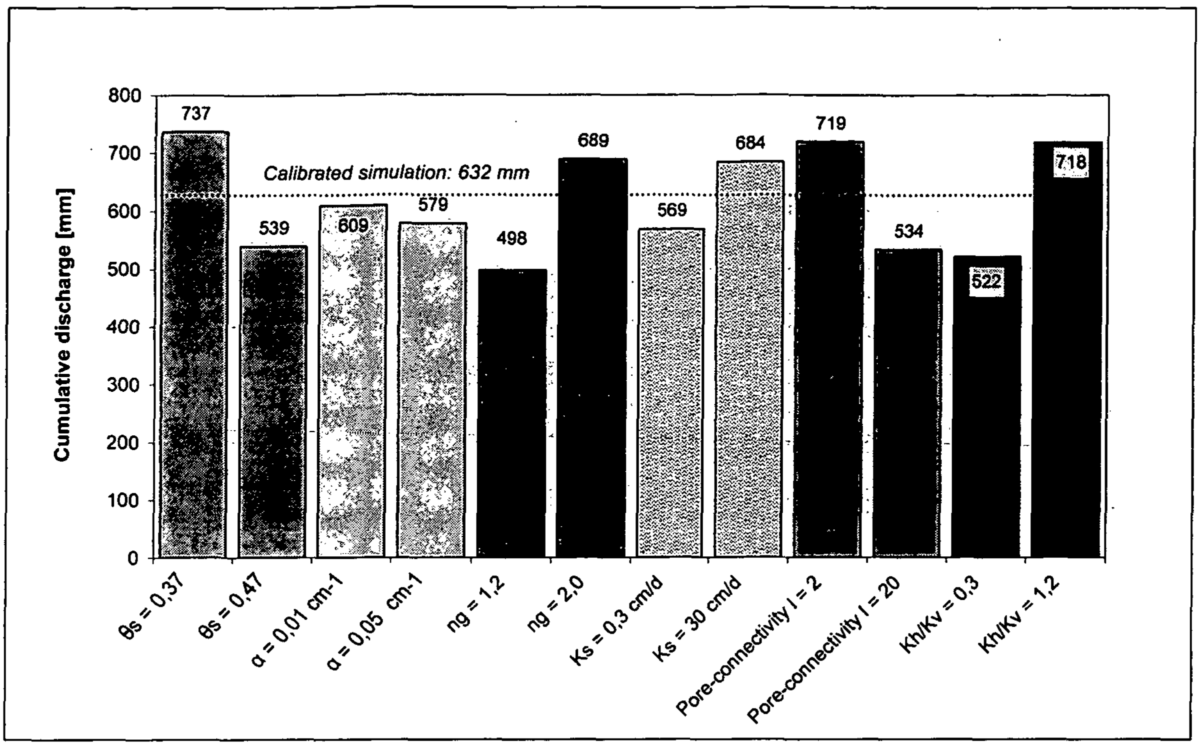



*Figure 5-32* ***Sensitivity analysis for model parameters of the matrix domain (cumulative discharge of Cell 2)***

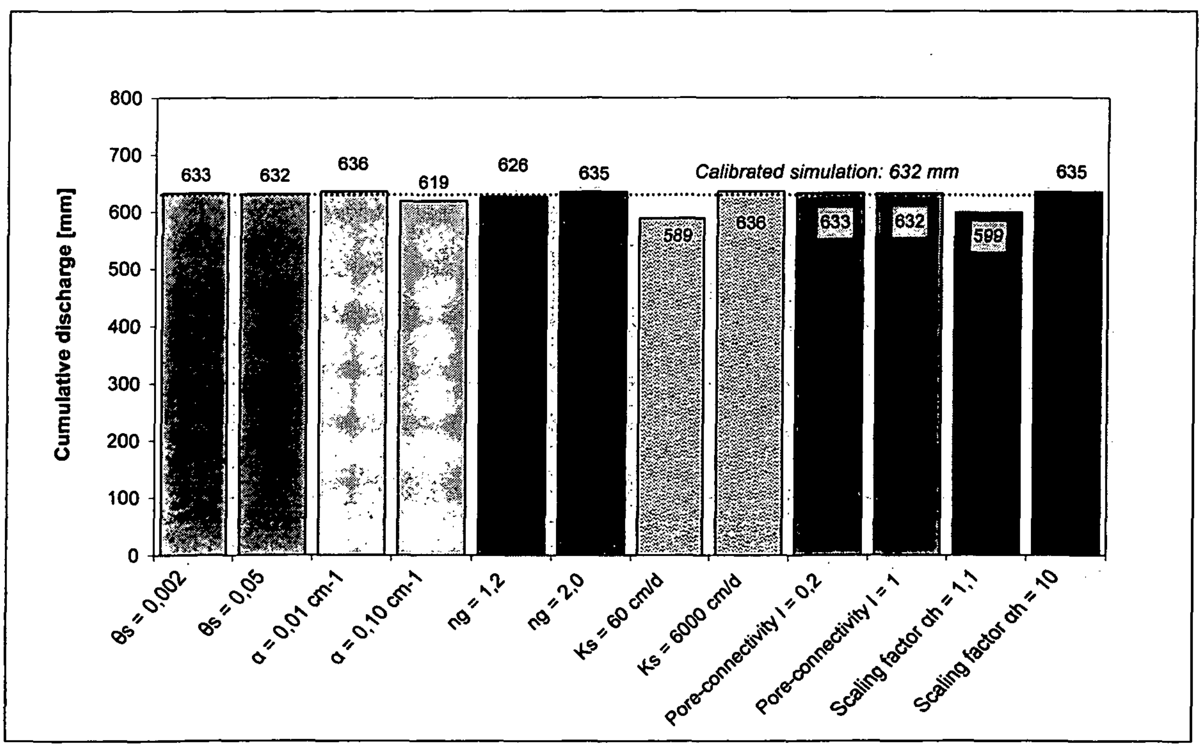



*Figure 5-33* ***Sensitivity analysis for model parameters of the channel domain (cumulative discharge of Cell 2)***

The sensitivity analyses showed that the results of the water flow model are mainly dependent on the hydraulic characteristics of the matrix domain. The variation of channel domain parameters had less impact on the results, with the exception of the saturated hydraulic conductivity $K_s$ and the pressure head scaling factor $\alpha_h$.

Total water storage, and thus cumulative leachate discharge is strongly affected by the following matrix domain parameters:

- residual $\theta_r$ and saturated water content $\theta_s$, respectively their difference,
- shape of the retention characteristics ascertained by the form parameters $\alpha$ and $n_g$, and
- hydraulic conductivity and its anisotropy given by the saturated hydraulic conductivity $K_s$, the pore-connectivity l and the anisotropy $K_h^A$ (ration between horizontal and vertical hydraulic conductivity $K_h/K_v$).

Of these parameters only the water contents $\theta_r$ and $\theta_s$ impact exclusively water storage processes. The other parameters influence also the temporal discharge characteristic.

The base flow from the landfill during dry periods is sensitive to the pore-connectivity l and the form parameter $n_g$ of the matrix domain. Additionally the saturated hydraulic conductivity $K_s$ of both domains and the pressure head scaling factor $\alpha_h$ (channel domain) affect to some extent the base discharge.

Shape and amplitude of leachate generation peaks caused by heavy rainfall are mainly dependent on the

- saturated hydraulic conductivity $K_s$ of both domains, respectively their ratio,
- pore-connectivity l of the matrix domain,
- anisotropy $K_h^A$ ($K_h/K_v$) of the hydraulic conductivity of the matrix domain, and
- pressure head scaling factor $\alpha_h$ of the channel domain.

Furthermore, the retention characteristics of the matrix domain given by the form parameter $n_g$ and $\alpha$ influences the amplitude of discharge peaks.

The sensitivity of the leachate hydrograph to the saturated hydraulic conductivity $K_s$ of the channel domain becomes minor after the parameter exceeds a "certain" threshold. The value of this threshold is affected by the hydraulic characteristics of the matrix domain. A similar effect is observable for the pressure head scaling factor $\alpha_h$.

In Table 5-20 the influence of single model parameters on water storage, on base flow during dry periods and on leachate discharge peaks is assessed, whereby the degree of influence is divided into three categories:

– = weak

\+ = medium

\+ + = strong

***Table 5-20 Influence of model parameters on the water flow through MSW landfills***

| | Parameter | *Water storage* | *Base flow* | *Discharge peaks* |
|---|---|---|---|---|
| Matrix domain | $\theta_r$ | + + | – | – |
| | $\theta_s$ | + + | – | – |
| | $\alpha$ | + | – | + |
| | $n_g$ | + + | + + | + |
| | $K_s$ | + | + | + |
| | l | + + | + + | + + |
| | $K^A_h$ ($K_h/K_v$) | + + | – | + + |
| Channel domain | $\theta_r$ | – | – | – |
| | $\theta_s$ | – | – | – |
| | $\alpha$ | – | – | – |
| | $n_g$ | – | – | – |
| | $K_s$ | + | + | + + |
| | l | – | – | – |
| | $\alpha_h$ | + | + | + + |

Summarizing the outcome of the sensitivity analysis, the results of the water flow model are mainly dependent on the hydraulic properties of the matrix domain. The characteristics of the channel domain primarily the parameter $K_s$ and $\alpha_h$ becomes decisive for water flow, if they fall below a "certain" threshold. Altogether nine parameters (seven for the matrix and two for the channel domain) influence the water flow, whereby depending on the considered "flow process" (water storage, base flow, discharge peaks) different parameters are crucial.

### 5.4. *Evaluation of flow regimes*

#### 5.4.1. Principles

The main aim for developing a new approach for modeling water flow through landfills is the need for a better insight into the hydraulic behavior of MSW landfills. Since water plays the key role in the metabolism of landfills, information on the transport of water is of direct interest for engineering stabilization processes or when evaluating the decomposition stage and predicting the duration of aftercare. Due to the highly heterogeneous nature of a landfill the flow field is not uniform. Internal structures of the landfill facilitate rapid water flow in restricted channel and voids, whereas large portions of the landfill are hardly participating in water flow (e.g. Zeiss & Major, 1993). This phenomenon is incorporated into the presented landfill model by dividing the waste mass into a matrix domain with slow water movement and a channel domain with fast water flow.

Sensitivity analyses for the model indicated that the following parameters influence the heterogeneity of the flow regime: $\alpha$, $n_g$, $K_s$, l, $K^A{}_h$ of the matrix domain and $K_s$, $\alpha_h$ of the channel domain. The values of these parameters can provide a first clue for assessing the uniformity respectively non-uniformity of the water flow. For instance large differences between the saturated hydraulic conductivity $K_s$ of channel and matrix domain, or high anisotropy $K^A{}_h$ of the hydraulic conductivity of the matrix domain result in heightened non-uniform water routing through the landfill.

Since the uniformity of the flow regime is influenced by seven hydraulic parameters, it is impossible to determine an overall quantitative hydraulic homogeneity grade for the water flow in landfills using the calibrated parameter values. However, such a hydraulic homogeneity grade quantifying the portion of waste mass participating in water flow would enhance insights into the metabolism of sanitary landfills. Current leachate emissions could be better evaluated regarding the stabilization stage of the whole landfill, since these reflect only the decomposition status of the preferential flow paths and their surroundings.

The calibrated hydraulic parameters of water flow simulations allow at the best only a qualitative evaluation of the water distribution. To obtain quantitative information about the portion of waste mass participating in water flow, knowledge on the flow velocity throughout the landfill is necessary. The simplest way to determine this characteristic is to perform solute transport simulations. In particular the discharge of conservative substances that are already dissolved in the leachate need to be modeled. In order to incorporate the hydraulic flow

regime of the considered landfill, the solute transport considerations must be based on the calibrated water flow model. The results of these simulations approximately represent the emission behavior of easy soluble salts (e.g. chloride or sodium) from the landfill.
Fortunately, the software HYDRUS-2D, on which the presented hydraulic landfill model is based, enables to simulate the transport of dissolved substances that are carried by water flow. The program uses the classical convection-dispersion-equation (Lapidus & Amundson, 1952) to compute the transport of solutes.

$$\frac{\partial c}{\partial t} = D\frac{\partial^2 c}{\partial z^2} - v\frac{\partial c}{\partial z}$$ ***Equation 5-1***

| | |
|---|---|
| $c$ | *Solute concentration [g m^-3]* |
| $t$ | *Time [s]* |
| $D$ | *Dispersion coefficient [m² s^-1]* |
| $v$ | *Pore water velocity [m s^-1]* |
| $z$ | *Coordinate [m]* |

In order to focus the transport processes to convective transport and thus water flow only, hydrodynamic dispersion (caused by different length of flow paths and different flow velocities within the pores) was neglected during the solute discharge simulations. However, some dispersion had to be accepted in order to avoid numerical instabilities of HYDRUS-2D (Šimunek & van Genuchten, 2002).
Theoretical solute transport considerations (solute discharge from a homogeneous porous media - Figure 5-34) regarding and disregarding hydrodynamic dispersion are show in Figure 5-35 (piston flow versus hydrodynamic dispersion). Additionally the effect of heterogeneous flow conditions on the discharge of dissolved salts is presented in Figure 5-35.

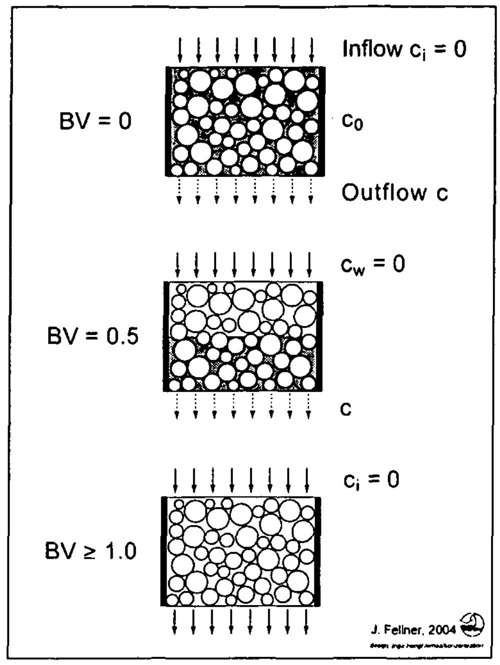



*Figure 5-34 Solute discharge from porous medium – sequence*

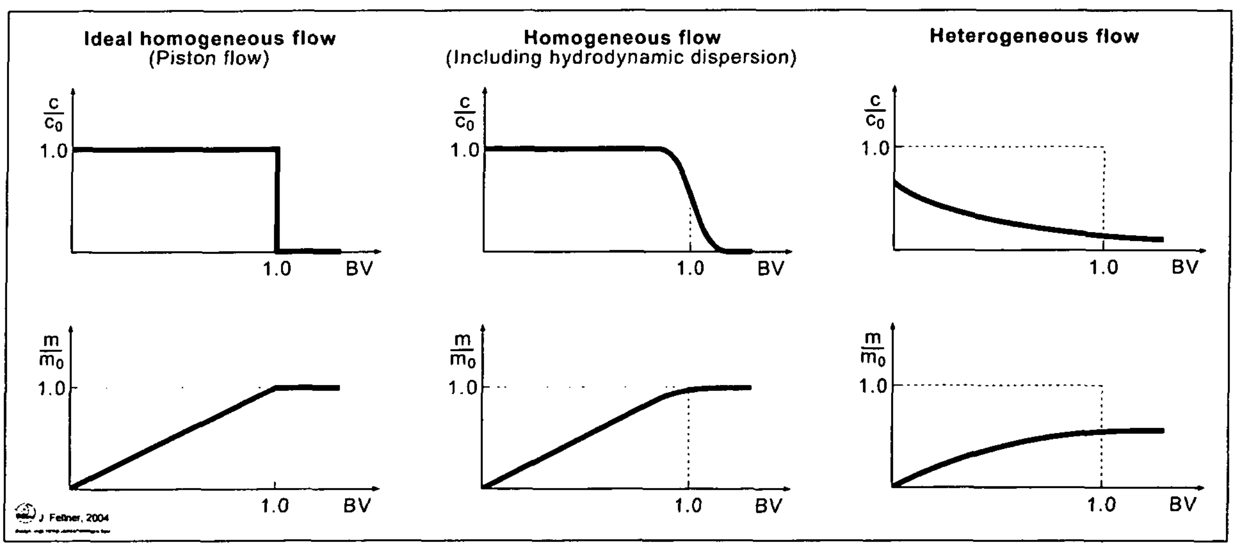



*Figure 5-35 Solute discharge (concentration and cumulative discharge) from porous media (piston flow, homogeneous and heterogeneous flow)*

$$BV = V_{tot} \cdot \theta \qquad \textit{Equation 5-2}$$

| | |
|---|---|
| $BV$ | *Bed volume* |
| $V_{tot}$ | *Total volume of the porous media* |
| $\theta$ | *Volumetric water content* |

| | |
|---|---|
| $c_o$ | *Initial solute concentration of the pore water* |
| $c$ | *Effluent concentration* |
| $c_i$ | *Inflow concentration (=0)* |
| $m_o$ | *Initial solute mass of the pore water* |
| $m$ | *Discharged solute mass* |

The assumption of ideal homogeneous flow (piston flow) leads to a complete discharge of the initial solute mass (dissolved in the pore water) after a water exchange of one bed volume (BV). One bed volume equals the total amount of water present in the porous media (see Equation 5-2). The effluent concentration c drops sharply to zero at this point.
Solute transport simulations regarding hydrodynamic dispersion show a smooth drop in concentration after one BV. Nevertheless, almost the total solute mass is discharged after a water exchange of one bed volume.
For heterogeneous flow conditions the effluent concentration drops already at the beginning of water input. After an exchange rate of one bed volume part of the initial solute load is discharged only.

Solute transport simulations with HYDRUS-2D were performed to determine the fraction of waste mass taking part in water flow (convective transport is significant higher than diffusive transport). It was assumed that the considered substance (any salt) is dissolved and uniformly distributed throughout the landfill. The simulations focused on water flow and its effect on the solute discharge. Thus, dissolution, adsorption and diffusion processes as well as the presence of immobile water have been disregarded. In order to incorporate a possible impact of the water application rate on the uniformity of water flow, the solute transport simulations were carried out using the average water input rate of the landfill considered.
The modeling resulted in the cumulative discharge of the substance (salt) versus time respectively applied water amount. Figure 5-36 gives the outcome for the investigated landfills, whereby both cumulative solute discharge and applied water amount are standardized to the initial solute mass and the water amount stored inside the landfill. This scaling leads to the presented graph of normalized cumulative solute discharge versus bed volumes. Since only convective transport of dissolved compounds has been considered, the normalized cumulative solute discharge corresponds to the fraction of waste mass participating in "convective" water flow.

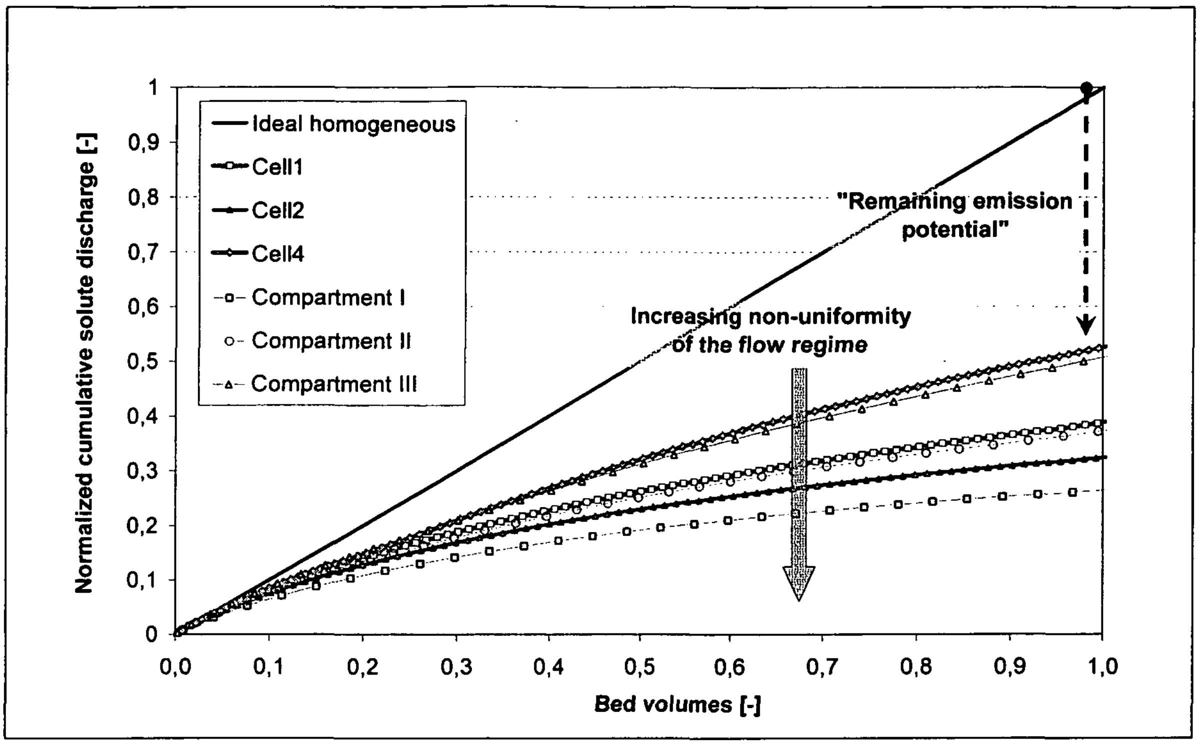



*Figure 5-36* ***Cumulative solute discharge (normalized) versus leachate flow (expressed in bed volumes) - comparison of all simulated landfills***

Figure 5-36 indicates that after one bed volume of water percolated through Compartment I of the Breitenau landfill only 26 % of the initial solute mass has been discharged. Under ideal homogeneous flow conditions (piston flow) the total amount of initial solute mass would have been flushed out after an exchange rate of one bed volume. The figure provides quantitative information on the heterogeneity of water flow. For Compartment I (Breitenau landfill) it can be stated that less than 30 % of the total landfill mass is participating in water flow (extrapolated graph), which means that even after high water exchange rates (corresponding to a long time period) more than 70 % of the initial pollution load (salt) is still remaining inside the landfill. Thus, observed leachate quality at Compartment I reflects only the decomposition stage of less than 30 % of the landfilled waste. Changes in the flow paths could lead to sudden increase in leachate concentration. That implicates that low concentration values at Compartment I which may be already "compatible" with the environment, cannot be used as indicators for the end of the aftercare period. The remaining emission potential must be taken into account to evaluate the stabilization status and thus, the end of aftercare measures. In general it can be stated that the aftercare period (time starting from landfill closure till the potential pollution load is removed) is extended by highly non-uniform water flow as prevailing at Compartment I. Additionally to the amount of water that passed the landfill, temporal changes of the water paths become significant for the stabilization of the whole waste mass.

### 5.4.2. Comparison of modeling results

A comparison of the results of all simulated landfills (Figure 5-36) demonstrate that water flow in Compartment I is most non-uniform, followed by test Cell 2 of the Spillepeng site and Compartment II of the Breitenau landfill. Water flow in Cell 1 is comparable to those in Compartment II. Most homogeneous water routing is observable at Cell 4 and Compartment III. There the portion of waste mass participating in water flow amounts more than 50 % after a water exchange of one bed volume.

Considering the discharge rates given in Figure 5-36 it must be kept in mind that easy soluble salts have been investigated. For substances undergoing biochemical decomposition (e.g. nitrogen) and/or adsorption it can be assumed that discharge rates are significant lower. For these substances different parameters can limit their discharge, since the degradation processes depend on various factors (water exchange, water content, pH, redox-potential, composite of nutrients, ...).

The results show that the information content of leachate quality regarding the stabilization stage of the whole landfill is best for Compartment III and Cell 4 compared to the other landfills that have been investigated. For these two landfills the collected leachate is representative for the decomposition stage of at least 50 % of the landfilled waste mass after a water exchange rate of one bed volume.

Significant differences in the non-uniformity of the water flow (Figure 5-36), as denoted for all three compartments of the Breitenau landfill, as well as for Cell 2 and Cell 4 of the Spillepeng test cells, are confirmed by investigations concerning leachate quality.

#### 5.4.2.1. Comparison of leachate quality (Breitenau landfill)

Figure 5-37 shows the development of the Sodium concentration versus cumulative discharge at the three compartments of the Breitenau landfill. The decrease of concentration values is highest for Compartment I, whereas Compartment III shows a relatively high concentration level of Sodium (1000 mg/l) after a total leachate discharge (water exchange) of

105 l/ton MSW. Identical cumulative water exchange at Compartment I results in Sodium concentrations of around 600 mg/l. Due to the fact that similar waste was landfilled, these significant differences in leachate concentration inevitably ascribe to diverse water routing. Slow decrease in leachate concentration as observed at Compartment III indicates a more uniform water flow, as a larger waste mass contributes to leachate pollution. Whereas rapid decrease of soluble compounds in the leachate (Compartment I) refers to preferential flow paths, that short a large bulk of waste. Low leachate concentration values in this case represent the favored flow paths and their surroundings only.

The heterogeneity of Compartment II lies according to its leachate characteristics in between the two other compartments. These results confirm the findings of the mathematical modeling (see Figure 5-36).

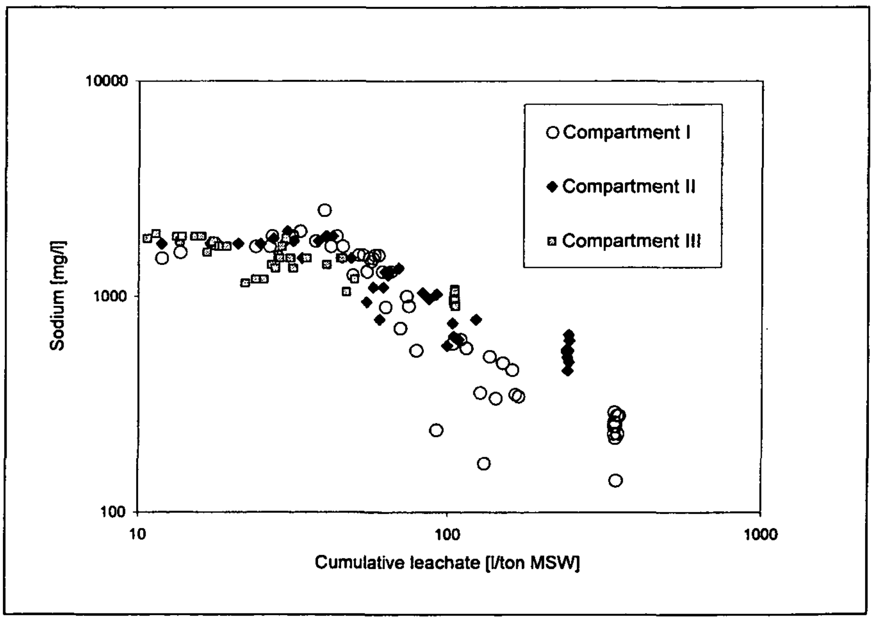



*Figure 5-37* ***Leachate quality (Sodium) versus cumulative leachate discharge (Breitenau landfill)***

### 5.4.2.2. Comparison of leachate quality (Spillepeng test cells)

The comparison of the flow regimes for Cell 2 and Cell 4 (Spillepeng test cells) had to be conducted differently due to insufficient records of the leachate concentration over the disposal time. Reliable information however, on the leachate quality over short periods (two years) was available. These data were used to investigate the influence of the discharge rate on the leachate concentration of easy soluble salts, since their emission behavior is only dependent on the water flow. The results of these investigations (Figure 5-38) denote that Chloride concentrations at Cell 2 are strongly influenced by the actual discharge rate. The dependency of the leachate quality at Cell 4 is significantly lower. For instance an increase of the leachate generation rate of one magnitude (0.1 to 1.0 mm/d) results for Cell 2 in a concentration decline of easy soluble compounds of around 50 %. For Cell 4 a reduction of only 20 % is observable. This fact can be seen as clear indication for diverse water routing in both cells. Uniform water movement in landfills is associated with slight influence of discharge rates on the leachate concentration, while the generation rate of leachate has major impact on its concentration in a flow regime of heightened heterogeneity. Consequently water flow in Cell 2 is more non-uniform compared to Cell 4. A result, that coincides with the outcome of the water flow modeling and thereon based solute discharge simulations.

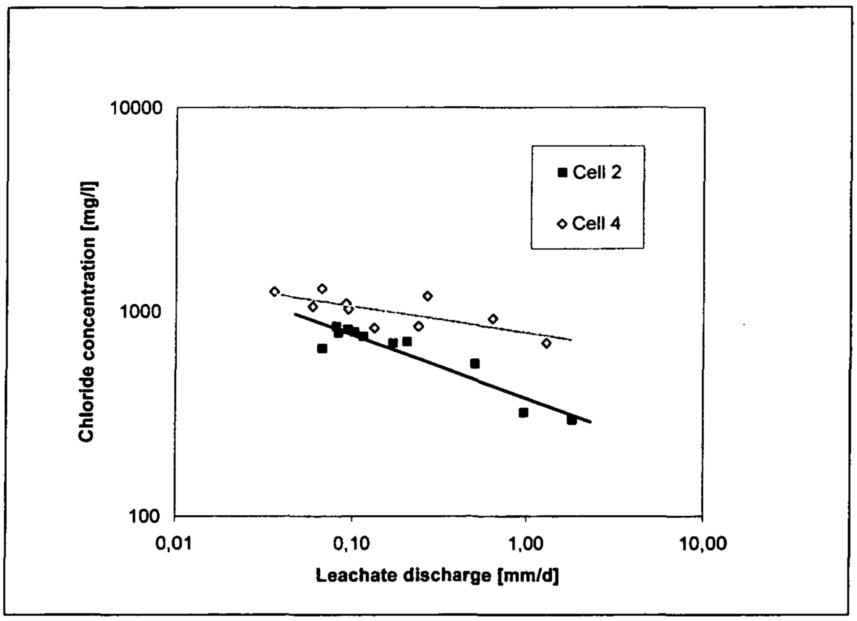



***Figure 5-38 Leachate quality (Chloride) versus leachate quantity (Cell 2 and Cell 4)***

The presented examples demonstrate the capability of the introduced landfill model for evaluating the homogeneity of the water flow inside a landfill. Quantitative information on the flow regime is essential for estimating the decomposition stage of the landfill using observed leachate quality. Also future emission behavior can only be predicted reliably, knowing the portion of the waste mass participating in water flow.

# 6. Summary and Conclusions

## *6.1. Introduction*

The main aim for developing a model for simulating water flow in MSW landfills was the need for a better insight into the black box "landfill". Since water plays the key role in the metabolism of landfills (e.g. Poland, 1975; Straub & Lynch, 1982b; Ehrig, 1983; Aragno; 1989) the mapping of water transport is of direct interest for engineering stabilization processes or when developing models for predicting leachate quality and biogas production.

## *6.2. Summary*

Up to now the prevailing approach for modeling water flow processes in solid waste media relied on the assumption of a homogeneous porous media (e. g. Straub & Lynch, 1982a; Schroeder et al., 1984; Korfiatis et al., 1984; Dematracopouls et al., 1986; Vincent et al. 1991; Noble & Arnold, 1991; Al-Yousfi et al., 1992; Demirekler et al., 1999). However, due to the heterogeneous nature of the waste media itself, the horizontal texture of the landfill caused by the landfilling and compaction technique, and construction elements with different hydraulic characteristics (such as gas wells or daily cover layers), the assumption of a homogeneous flow regime may be questioned. In several field investigations (e.g. Wiemer, 1982; Blight et al., 1992) it has been shown that the water content varies from saturated conditions to complete dryness inside the landfill. Preferential flow paths that short a large bulk of the landfill explain this. According to Ehrig (1983) and Uguccioni & Zeiss (1997) the rapid flow in those favored flow paths is believed to be the reason why existing landfill models are not in agreement with actual field observations. Also, the spatial and temporal variations in the leachate composition reported in the literature (El-Fadel, 1997; Åkesson & Nilson, 1997, Döberl et al., 2002) may partly attribute to non-uniform water flow.

In recently developed water flow models for MSW (Uguccioni & Zeiss, 1997; Bendz, 1998; Obermann, 1999) the heterogeneous character of landfills was taken into account. The waste body was not considered as a homogeneous media anymore. It was split into a channel domain with rapid water flow surrounded by a matrix domain with slow water movement. The mathematical approaches for describing the water flow in the two domains are different. Uguccioni & Zeiss (1997) used the model PREFLO (Workman & Skaggs, 1990) to simulate the water movement. This model assumes that the rapid flow in the channel domain follows Poiseuille's Law (1841) and the lateral water transfer from the channels into the matrix occurs according to Richards' Law (1931). Bendz (1998) used another assumption for describing the fast water flow in channels. A power function (kinetic wave model), as it has already been proposed by Beven & German (1981) to describe the water flow in macroporic soils, was used to determine the channel flow in landfills. Water filtrates into the matrix domain under wet conditions and is released again during dry periods. Obermann (1999) suggested a two-domain approach with Darcy flux in both zones, in the matrix as well as in the channel domain. Fast channel flow occurs only if the water input exceeds the hydraulic conductivity of the matrix domain. The application of two-domain water flow models partly results in a better fit between predicted and observed leachate generation rates. However, a large number of unknown model parameters must be accepted using these approaches. Up to now simulations are limited to laboratory experiments only. A framework for scaling up and validating models at landfill size is lacking. This is mainly due to sophisticated mathematical formulations that complicate the incorporation of variable boundary conditions prevailing at landfill sites enormously. Furthermore, all introduced two-domain concepts have in common that they are derived from the framework that has been carried out for non-uniform water flow in soils. Compared to soils, landfills are more heterogeneous, which inevitably results in a bigger fraction of preferential flow. This fact was taken into account in existing model approaches by adjusting the decisive parameters. However, a comparison of the characteristics of non-uniform water movement in cracked soils and MSW landfills point out significant differences. Whereas in soils preferential flow becomes minor with increasing depth (Bundt et al., 2000) the opposite effect is observable in landfills, represented by accelerated transport of solutes (Rosqvist et al., 1997) and bigger variation in water content towards the landfill bottom (e.g. Yuen et al., 2001). Moreover, rapid flow in fissures and macro-pores of soils is limited to wetting periods, while landfills show significant channel flow even during dry periods.

The disregard of these basic differences in the water flow pattern of soils and MSW landfills led to the development of a new two-dimensional two-domain approach. The proposed water flow pattern (Figure 6-1), derived from findings of different landfill studies, has been realized using the software tool HYDRUS-2D (Šimunek et al., 1996). This model enables to simulate water flow, solute and heat transport in variably saturated porous media.

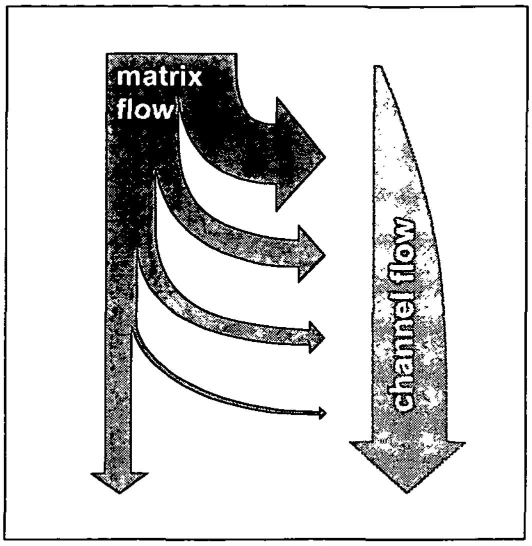



*Figure 6-1 General flow pattern of water in MSW landfills*

A two-dimensional flow field consisting of one vertical favored flow path surrounded by the waste mass (matrix domain) was defined using HYDRUS-2D. The preferential flow path (channel domain) was assigned a high permeability and a low or even no-retention capacity, while the matrix domain is characterized by low permeability and high retention capacity. Water flow in both domains is calculated according to Richards equation, whereby "virtual" suction power within the preferential flow path is assumed to ensure the proposed flow pattern. HYDRUS-2D accounts for variable boundary conditions (e.g. precipitation and evapotranspiration), which facilitates its application to field data. Runoff processes are not incorporated and must be considered separately.

Simulation results of the developed model concept were presented for two landfill sites: the experimental landfill Breitenau in Austria (~ 95,000 tons of waste), and the Spillepeng landfill in Malmö, Sweden (~ 25,000 tons of waste). Altogether leachate generation from six different waste compartments was simulated. The model results showed a good match with observed leachate generation rates. However, it was necessary to calibrate numerous parameters (14). Initial estimates of the parameters to calibrate were derived from hydraulic investigations at MSW landfills reported in the literature. By means of sensitivity analysis five parameters crucial for the overall storage of water inside the landfill were identified. These variables (residual and saturated water content: $\theta_r$ and $\theta_s$, form parameter $n_g$ of the retention characteristics, pore-connectivity l and anisotropy $K^A_h$ of the hydraulic conductivity) characterize the hydraulic properties of the matrix domain only. Moreover, it was shown that also the non-uniformity of the water flow is mainly dependent on the characteristics of the matrix domain (anisotropy $K^A_h$, pore-connectivity l, form parameter $n_g$). Additionally to these parameters, the difference of the saturated hydraulic conductivities $K_s$ between both domains as well as the scaling factor for the pressure head $\alpha_h$ ("virtual suction power") have major impact on the shape of discharge hydrographs, and thus, the heterogeneity of the water flow in landfills.

Quantitative information on the uniformity of the water flow in landfills was obtained by solute transport simulations that are based on the calibrated water flow model. Thereto the discharge of a conservative substance was computed using HYDRUS-2D. In order to focus on water flow only, the solute discharge modeling was restricted to convective transport. The simulations result in graphs that provide the fraction of the initial solute load that has been discharged over time. This corresponds to the portion of waste mass participating in water flow versus cumulative water exchange rate (see Figure 6-2).

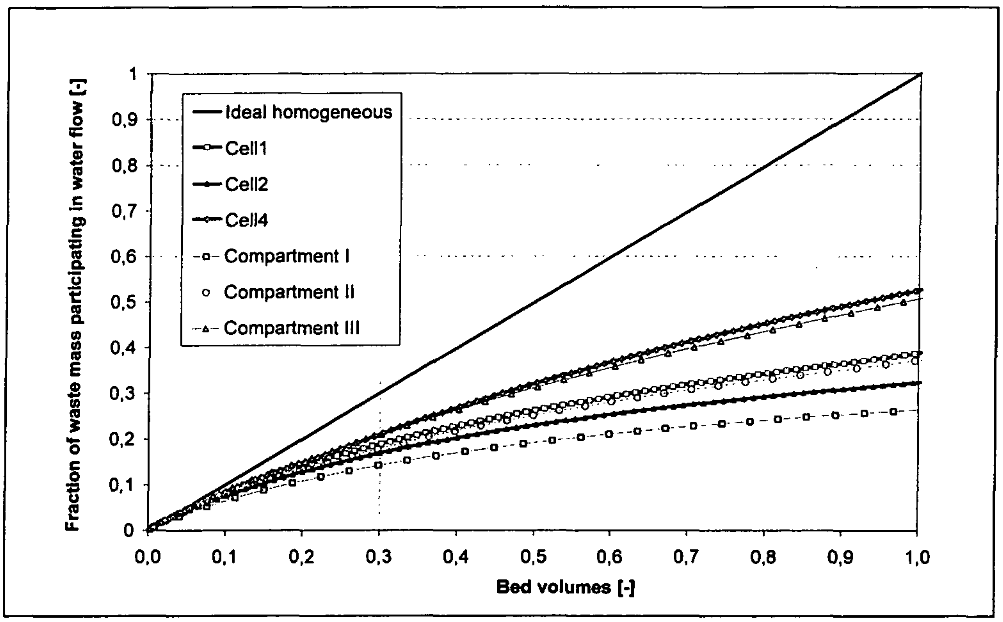



***Figure 6-2*** ***Fraction of waste mass participating in water flow versus water exchange (expressed in bed volumes)***

The results of the solute discharge simulations (Figure 6-2) indicate that the water flow in Compartment I (Breitenau landfill) is highly non-uniform, whereas Cell 4 shows the most uniform water flow of the investigated landfills. However, at most 50 % of the waste mass is participating in water flow after an exchange rate of one bed volume. Although, similar waste was landfilled and compacted in the same way (for each case: Breitenau landfill and Spillepeng test cells), significant differences in the uniformity of the water transport have been identified. Different water flow in the considered landfills results obviously in unlike metabolisms. Degradation processes of the waste mass excluded from water exchange are strongly decelerated (e.g. Klink & Ham, 1982; Bogner & Spokas, 1993), as water is the only carrier of substances within a landfill and only water flow facilitates the redistribution of chemicals, micro-organisms and nutrients.

The new model concept enables to quantify the hydraulic homogeneity of landfills, which leads to a better understanding of the metabolism and future emission behaviour of sanitary landfills.

When applying the software HYDRUS-2D for simulating water flow and solute transport, it was noticed that the model shows numerical problems for highly non-uniform flow regimes. Nevertheless, numerical instabilities could be avoided defining small meshed grids (spacing of less than 10 cm), in particular at the interface between matrix and channel domain.

## *6.3. Conclusions*

The differences in water flow pattern of heterogeneous soils and MSW landfills are not only important for mathematical modeling, but also for landfill engineering. The operation of modern landfills as "flushing bioreactor" is recommended by several researchers (e.g. Gronow, 1993; Reinhart & Townsend, 1998; Beaven & Knox, 1999). The purpose is to achieve a stable landfill within one generation (30 years). Enhancement of biochemical degradation processes as well as flushing of easily soluble compounds is the main objective of this strategy. However, preferential pathways that short a large bulk of waste mass must be accounted for when applying this method. It is shown in seceral investigations that water flow becomes more non-uniform towards the landfill bottom (e.g. Rosqvist et al., 1997). Thus, an evenly two-dimensional water application directly underneath the landfill cover, as promoted by Drees (2000), results in less waste exposed to the flushing water. In order to increase the participating water volume it is suggested to inject water in different depths, whereby an augmented number of feeding points is needed in bigger depths. Despite better insights into the landfill reactor, and probably improved water feeding, the operation strategy of flushing bioreactor must be questioned due to the huge water consumption, and thus, the enormous costs for leachate treatment even if part of the leachate is recirculated.

Existing approaches for assessing the remaining pollution potential and the future emission behavior of MSW landfills use either the observed actual leachate quality or the amount of water passed through the landfill (e.g. Allgaier & Stegmann, 2003) as the main indicator for the decomposition stage of landfills. However, as a major part of water flowing in landfills is restricted to favored flow paths, thereby bypassing large bulk of waste, observed leachate quality reflects only the flow paths and their surroundings. Sudden changes in the physical structure of the landfill (e.g. settlement due to biodegradation) may change the water routes. Thus, new parts of the landfill may be exposed to moving water. Consequently, the quality of the leachate may increase. Also the stabilization indicator "water amount passed through the landfill" (liquid to solid ratio) neglects the effect of preferential flow, associated with zones of high water exchange, and zones of nearly no exchange with high remaining pollution potential. In order to improve prediction of leachate quality and assessment of remaining pollution potential the magnitude of the true volume participating in the water flow through a landfill must be evaluated. The introduced landfill model based on HYDRUS-2D enables to estimate this volume by solute transport simulations for easy soluble salts. The model results show that the fraction of waste mass taking part in water flow varies considerably even in landfills of similar waste. Observed leachate qualities at landfill sites can be evaluated better regarding the degree of stabilization taking the prevailing flow conditions into account. For instance, equal leachate concentration levels could denote highly different decomposition stages of landfills (e.g. Compartment I versus Cell 4), since different fractions of the total waste mass may be reflected by the leachate. Also, existing assessment tools for landfill stabilization that are based on cumulative leachate quantity can be advanced by incorporating information on the water flow. The results for landfills with highly non-uniform water flow (e.g. Compartment I, see ) indicate that temporal changes of the water paths become important for the duration of the aftercare.

In addition to a better assessment of the landfill stabilization the new model allows comparing the hydraulic homogeneity of different landfills. Also the uniformity of water flow in small-scale experiments (e.g. landfill simulation reactors) can be determined using this method. When comparing the fraction of waste mass participating in water flow in full size landfills and small waste columns, the capability of laboratory experiments for predicting leachate generation can be assessed.

Practical examples for the application of the new model will be provided in an ongoing research project "A New Method to Characterize the Stability of Old, Large Size Landfills" conducted in cooperation between the Institute of Water Quality and Waste Management, Vienna University of Technology and the Department of Waste Management, Technical University of Hamburg-Harburg.

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# 8. Appendix

## *8.1. Short Description of HYDRUS-2D*

(after Bonaparte et al., 2004)

HYDRUS-2D is a two-dimensional unsaturated flow model developed at the U.S. Salinity Laboratory (Šimůnek et al., 1999). The model also simulates heat flow and solute transport. The current model is an extension of the earlier unsaturated flow codes SWMS_2D and CHAIN_2D. At the time of this writing version 2.02 of HYDRUS-2D was the most current. The model may be purchased from the International Ground Water Modeling Center, Colorado School of Mines, Golden, Colorado or http://www.Mines.EDU/research/igwmc/software/igwmcsoft/. The documentation and a free demo version of HYDRUS-2D may be downloaded from http://www.ussl.ars.usda.gov/models/hydrus2d/htm.

HYDRUS-2D uses a finite element method to solve Richards' equation in a plane oriented either vertically or horizontally. The two-dimensional domain may take on any geometric shape. Because the model is two-dimensional, lateral flow and anisotropy may be simulated. A sink term is included in Richards' equation for removal of water via plant transpiration. Vapor flow cannot be simulated. The model has an option for allowing soil properties to be temperature dependent, and it also allows hysteresis and spatial variability through a scaling transformation (Vogel et al., 1991). The unsaturated hydraulic conductivity is calculated by either a Brooks-Corey, van Genuchten-Mualem, or modified van Genuchten method. Precipitation, runoff, ET, soil water storage, and percolation are included in the water balance. Precipitation and potential evaporation are the only climatic inputs required. HYDRUS-2D does not have an option for internally calculating potential evaporation, so the user must use another model or method to generate data to input. Vegetation parameters required include the heads between which transpiration occurs and also the heads between which transpiration is optimal. A menu containing a variety of properties for plants is available. The distribution of roots must also be specified. Input required for soil properties includes saturated hydraulic conductivity and fitting parameters from the selected soil-water retention function. A menu of soil properties is available. In addition, van Genuchten properties can be predicted by inputting the percentage of sand, silt and clay, density, field capacity, and/or wilting point water content. HYDRUS-2D also has the option for inverse estimation of soil hydraulic properties from measured flow data.

The two-dimensional profile is created through a pre-processing module called Meshgen2D within the HYDRUS-2D graphical user interface. After the domain geometry is defined, Meshgen2D assists in generating the finite element mesh.

Boundary conditions may be specified flux, specified pressure head, unit gradient, atmospheric, seepage face, or deep drainage. Precipitation and potential evaporation are specified using the atmospheric option, which allows the boundary condition at the soil surface to change from either prescribed flux or prescribed head. The user inputs the upper and lower limits of head for which the prescribed flux boundary operates. Therefore, evaporation and precipitation will proceed at the potential rate until the soil surface dries or wets to a specified head. Once below the specified head, the boundary changes to a prescribed head boundary condition, and evaporation is limited by the ability of water to flow to the surface. If the surface becomes saturated during precipitation, excess precipitation is removed as runoff. The seepage face option allows water to exit the domain when the soil adjacent to the boundary becomes saturated. Deep drainage provides an option for a variable flux depending on the level of the groundwater table. Initial conditions may be specified as either water contents or pressure heads.

The HYDRUS-2D post-processor allows a variety of options for viewing output. Results can be displayed graphically, including an animation of changes in pressure head or water content through time. Cross-sections plotting pressure head or water content vs. depth or length may be taken from the profile at any time of the simulation. Other output options include viewing the instantaneous or cumulative water boundary fluxes over time, run time information, graphical display of soil hydraulic properties, or converting output to ASCII format.

## 8.2. *Retention models*

van Genuchten model (1980):

$$\theta(h) = \theta_r + \frac{\theta_s - \theta_r}{\left[1 + (\alpha \cdot h)^{n_g}\right]^{1-1/n_g}}$$ **Equation 8-1**

$$S_e = \frac{\theta(h) - \theta_r}{\theta_s - \theta_r} = \left[1 + (\alpha \cdot h)^{n_g}\right]^{1-1/n_g}$$ **Equation 8-2**

$$K(S_e) = K_s \cdot S_e^{\,l}\left\{1 - \left[S_e^{\,n_g/(n_g-1)}\right]^{1-1/n_g}\right\}^2$$ **Equation 8-3**

Modified van Genuchten model (Vogel & Cislerova, 1988)

$$\theta(h) = \begin{cases} \theta_a + \dfrac{\theta_m - \theta_a}{\left(1 + |\alpha \cdot h|^{n_g}\right)^m} & h < h_s \\ \theta_s & h \geq h_s \end{cases}$$ **Equation 8-4**

Brooks and Corey (1964)

$$S_e = \frac{\theta(h) - \theta_r}{\theta_s - \theta_r} = (\alpha \cdot h)^{-\lambda}$$ **Equation 8-5**

$$K(S_e) = K_s \cdot S_e^{\frac{2+3\lambda}{\lambda}}$$ **Equation 8-6**

Campell (1974) und Huston & Cass (1987)

For $\theta \leq \theta_c$: $$h = a\left(\frac{\theta}{\theta_s}\right)^{-b}$$ ***Equation 8-7***

For $\theta \geq \theta_c$: $$h = \frac{a\left(1-\frac{\theta}{\theta_s}\right)^{1/2}\left(\frac{\theta_c}{\theta_s}\right)^{-b}}{\left(1-\frac{\theta_c}{\theta_s}\right)^{1/2}}$$ ***Equation 8-8***

whereby $$\theta_c = \frac{2b\theta_s}{1+2b}$$ ***Equation 8-9***

$$h_c = \frac{2b}{1+2b}$$ ***Equation 8-10***

$$K(\theta) = K_s\left(\frac{\theta}{\theta_s}\right)^{2b+2+p}$$ ***Equation 8-11***

$\theta$ *volumetric water content [$m^3$ $m^{-3}$]*

$\theta_s$ *saturated volumetric water content [$m^3$ $m^{-3}$]*

$\theta_r$ *residual volumetric water content [$m^3$ $m^{-3}$]*

$h$ *suction (pressure) head [m]*

$\alpha$ *form parameter [1/m], inverse bubbling pressure*

$n_g$ *form parameter [-]*

$K$ *hydraulic conductivity [m $s^{-1}$]*

$K_s$ *saturated hydraulic conductivity [m $s^{-1}$]*

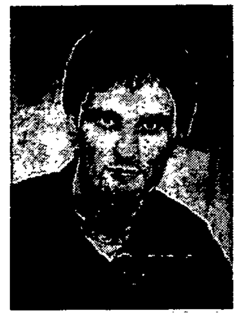
$l$ *pore-connectivity parameter [-]*

$S_e$ *degree of saturation [-]*

$\theta_a$ *$\leq \theta_r$ extrapolated volumetric water content [$m^3$ $m^{-3}$] at infinite small matrix potential*

| | |
|---|---|
| $\theta_m$ | *$\geq \theta_s$ extrapolated volumetric water content [$m^3\ m^{-3}$], at full water saturation* |
| $\theta_r$ | *volumetric water content [$m^3\ m^{-3}$] at the hydraulic conductivity $K_k$* |
| $\lambda$ | *empirical pore coefficient [-]* |
| $K_k$ | *hydraulic conductivity [$m\ s^{-1}$] at a water content of $\theta r$* |
| $L$ | *empirical tortuosity coefficient according to Mualem (usually 0.5)* |
| $a$ | *air entry value [m]* |
| $b$ | *Campell exponent [-]* |
| $p$ | *pore coefficient [-]* |

# CURRICULUM VITAE



| | |
|---|---|
| Name: | Johann Fellner |
| Date of birth: | 06.03.1974 |
| Place of birth: | Neunkirchen, Austria |

**Education:**

| | |
|---|---|
| *2001 – 2004* | Doctoral student, Vienna University of Technology, Faculty of Civil Engineering, Institute for Water Quality and Waste Management |
| *1993 – 1999* | Master student, Land and Water Management and Engineering, Vienna University of Natural Resources and Applied Life Sciences. |
| | Master thesis: "Comparison of Solute Transport Models", Institute for Hydraulics and Rural Water Management |
| *1988 – 1993* | College for Engineering Wr. Neustadt, Electrical Engineering |
| *1980 – 1988* | Primary School St. Lorenzen, and Comprehensive Secondary School Neunkirchen |

**Work experience:**

| | |
|---|---|
| *Since 07/2003* | Senior research associate, Vienna University of Technology, Institute for Water Quality and Waste Management |
| *01 – 06/2003* | Visiting Researcher, Department of Water Resources Engineering, Lund University, Sweden |
| *10/2000 – 12/2003* | Research assistant, Vienna University of Technology, Institute for Water Quality and Waste Management |
| *10/1999 – 09/2000* | Civil service at the Neunkirchen hospital |
| *1998 – 1999* | Engineering work in the field of water and waste management, ÖKOTEC, Wr. Neustadt |

**DISSERTATION**

# A New Method for Modeling Water Flow and Water Storage in Municipal Solid Waste Landfills

## Eine neue Methode zur Modellierung der Wasserströmung und Wasserspeicherung in Hausmülldeponien

ausgeführt zum Zwecke der Erlangung des akademischen Grades eines Doktors der technischen Wissenschaften unter der Leitung von

o.Univ. Prof. Dr. Paul H. Brunner
Institut für Wassergüte und Abfallwirtschaft 226

eingereicht an der Technischen Universität Wien
Fakultät für Bauingenieurwesen

Dipl.-Ing. Johann Fellner
Matr.Nr. 9340369
1130 Wien

Wien, Mai 2004

*"Nessuna umana investigazione si può dimandare vera scienza, s'essa non passa per le matematiche dimostrazioni"*

Leonardo da Vinci (1452 – 1519)

*„No human investigation can be entitled true science if it does not proceed by the way of mathematical demonstration"*

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Not be forgotten are my parents. Their support kept me going throughout my studies, and I would not have been able to complete everything without them.

Finally, I would like to express my gratitude to my beloved Sahar for her devotion and support during this study.

## ABSTRACT

This thesis focuses on the numerical modeling of water movement and water storage in municipal solid waste (MSW) landfills. In particular the temporal and spatial leachate generation is considered. Hydraulic investigations at landfill sites indicate that the water flow is highly non-uniform. Preferential flow paths dominate the water transport. The non-uniform flow regime is caused by the heterogeneous character of the waste material itself, the disposal and compaction procedure, and by the construction elements such as gas wells or daily cover layers. Landfill models that incorporate preferential flow originate from soil physics and were developed for fissured or cracked soils. However, the special textural characteristics of landfills lead to different water flow patterns. Contrary to soils, water flow in landfills is funneled to favored pathways with increasing depth. Moreover, preferential flow in landfills occurs also during dry periods.

In this thesis a two-dimensional two-domain approach for modeling water flow in landfills has been developed. Thereby a flow field consisting of one vertical favored flow path (channel domain) surrounded by the waste mass (matrix domain) is defined using the software HYDRUS-2D. This model enables the calculation of water flow, solute and heat transport in porous media at variable boundary conditions. The results show that water in landfills follows a preferential path determined by high permeability and low or even no retention capacity. The bulk of the landfill (matrix domain) is characterized by low permeability and high retention capacity.

The water flow model is calibrated using data from two landfill sites in Austria (Breitenau) and Sweden (Spillepeng). Predicted leachate generation corresponds well with the observed discharge. Parameters calibrated and thus heterogeneity of the flow regime is different for the two landfills. In order to quantify the heterogeneity of the flow regime, the transport of highly soluble salts is investigated. The calibrated water flow model and HYDRUS-2D were used to simulate the solute discharge. This allows determining the fraction of waste mass engaged in water flow. For the investigated landfills this fraction varies between 25 % and 50 %.

The new model improves prediction of future emissions of MSW landfills, because it allows assessing flows and stocks of water, the key variables in landfills, in a quantitative way.

## DEUTSCHE KURZFASSUNG

Ziel der Arbeit ist die Entwicklung eines mathematischen Modells zur Beschreibung der Wasserbewegung und Wasserspeicherung in Hausmülldeponien. Das Hauptaugenmerk liegt in der Bestimmung des örtlichen und zeitlichen Sickerwasseraufkommens.

Hydraulische Untersuchungen an Hausmülldeponien zeigen, dass die Wasserverteilung innerhalb des Deponiekörpers heterogen ist. Präferenzielle Fliesswege bestimmen das Abflussgeschehen. Große Teile der Deponie sind kaum Wasser durchflossen. Verantwortlich für die ungleichmäßige Wasserverteilung sind die unterschiedlichen Materialeigenschaften und die spezielle Struktur des Deponiekörpers (lagenweiser Einbau des Abfalls, Konstruktionselemente wie Gasbrunnen und Zwischenabdeckungen). Bisherige Deponiemodelle in denen präferenzieller Abfluss berücksichtigt wird, sind an Wasserhaushaltsmodelle für Böden angelehnt. Ein Vergleich von Böden und Deponien zeigt jedoch wichtige Unterschiede. In Böden nimmt der Anteil an präferenziellem Abfluss mit zunehmender Tiefe ab, in Deponien hingegen zu. Zusätzlich erfolgt in Deponien konträr zu Böden ein Abfluss über bevorzugte Sickerwege auch während Trockenwetterperioden.

Diese besonderen Strömungsverhältnisse wurden in einem neuen zwei-dimensionalen zwei-Bereichsansatz berücksichtigt. Der Deponiekörper wird in einen feinporigen Matrixbereich mit geringer hydraulischer Durchlässigkeit und hohem Speichervermögen sowie einen vertikalen Sickerpfad mit hoher Durchlässigkeit und vernachlässigbarer Speicherkapazität unterteilt. Die mathematische Umsetzung dieses Konzeptes erfolgt mit Hilfe des Stofftransportmodells HYDRUS-2D. Dieses Programm ermöglicht es den Wasser-, Stoff- und Wärmetransport in variabel gesättigten porösen Medien unter Berücksichtigung veränderlicher Randbedingungen zu berechnen. Das Modell wird anhand von Messdaten zweier Deponien in Österreich (Breitenau) und Schweden (Spillepeng) kalibriert. Mit dem zweidimensionalen zwei-Bereichsansatz kann eine gute Übereinstimmung zwischen gemessenen und berechneten Sickerwasserabfluss erreicht werden. Unterschiede bei den kalibrierten Parameterwerten für die einzelnen Deponien können durch unterschiedliche Wasserverteilung in den Ablagerungen erklärt werden. Um ein Maß für die Homogenität der Wasserströmung zu erhalten, wird mit Hilfe von HYDRUS-2D der Austrag von leicht löslichen Salzen modelliert. Für die zwei Deponien variiert der von Wasser durchströmte Anteil zwischen 25 % und 50 %. Selbst für Deponieabschnitte, die mit ähnlichem Abfall und auf dieselbe Weise verfüllt wurden, sind erhebliche Unterschiede in der Homogenität der Wasserströmung feststellbar.

Da im Deponiekörper ablaufende Reaktionen stark vom Wassergehalt und Wasseraustausch abhängig sind, ermöglicht das entwickelte Modell anhand der Kenntnisse der Strömungsverhältnisse und des Wasserdurchsatzes den Stabilisierungsgrad und damit das zukünftige Emissionsgeschehen von Hausmülldeponien besser abschätzen zu können.

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## LIST OF ACRONYMS

| | |
|---|---|
| BV | Bed volume |
| BTC | Breakthrough time |
| CDE | Convection-dispersion-equation |
| CI | Compartment I |
| CII | Compartment II |
| CIII | Compartment III |
| COD | Chemical oxygen demand |
| FILL | Flow investigation for landfill leachate |
| HELP | Hydrologic evaluation of the landfill performance |
| L/S | Liquid to solid ratio |
| LSR | Landfill simulation reactor |
| MSW | Municipal solid waste |
| PTF | Pedo-transfer-function |
| PVC | Polyvinylchloride |
| SCS | Soil conservation service |
| SUTRA | Saturated-unsaturated flow and transport model |
| TOC | Total organic carbon |
| WS | Wet substance |
| ZAMG | Zentralanstalt für Meteorologie und Geodynamik (Central Institute of Meteorology and Geodynamics) |

## GLOSSARY

*Base flow:* The portion of the discharge that is derived from storage

*Bed volume:* The amount of pore space occupied by water

*Breakthrough curve*: The relative solute concentration in the outflow from a column of a porous medium after a step change in solute concentration has been applied to the inlet end of the column, plotted against the volume of outflow (often in number of bed volumes).

*Channel domain:* The domain representing connected fissures and preferential pathways with a pore diameter > 50 μm

*Conservative substance:* A substance whose concentration in water does not change, except by dilution (does not undergo adsorption or degradation processes)

*Field capacity:* The volumetric water content in a porous medium 2 - 3 days after being saturated (by rainfall or irrigation) and after free drainage has ceased

*Fraction of waste mass participating in water flow:* In this portion of the landfill the convective solute transport is significant (at least one magnitude) higher compared to diffusive transport (that implicates a water flux density > 0.05 mm/d)

*Gravitational Potential:* The gravitational potential of water is the amount of work required per unit quantity of water to move a very small amount of water reversibly and isothermally from a pool of pure water at atmospheric pressure at a reference level to another pool of pure water at the elevation of interest.

*Hydraulic flow regime:* The magnitude, timing, duration, distribution and frequency of water flow

*Hydraulic homogeneity:* (= uniformity of water flow) At each point within the porous medium the water flux density is constant

*Hydrodynamic dispersion*: The process wherein the solute concentration in flowing solution changes in response to the interaction of solution movement with the pore geometry of the porous media, a behavior with similarity to diffusion but only taking place when solution movement occurs

*Hydraulic homogeneity grade:* Measure for the hydraulic homogeneity expressed by the fraction of waste mass participating in water flow

*Matrix domain:* The domain representing the fine pored bulk of waste with a maximum pore diameter of 50 μm

*Matrix Potential:* The matrix potential of water in a porous medium is the amount of work required per unit quantity of water to move a very small amount of water reversibly and isothermally to the point of interest in the porous medium from a pool of pure water at atmospheric pressure at the same elevation.

*Pore water velocity:* The velocity at which water travels in pores relative to a given axis. It is equal to the water flux density divided by the volumetric water content

Preferential flow: The process whereby free water and its constituents move by preferred pathways through a porous medium

*Stabilization:* The reduction of the emission potential of the landfilled waste with the objective of final storage quality

*Total Potential:* The total potential of water in a porous medium is the amount of work required per unit quantity of water to move a very small amount of water reversibly and isothermally from a pool of pure water at atmospheric pressure and at a reference level to the point of interest in the porous medium. This is the sum of the matrix potential and the gravitational potential

*Uniformity of water distribution:* At each point within the porous medium the volumetric water content is constant

*Water flux density:* The volume of water passing through the porous medium per unit cross-sectional area (perpendicular to the flow) per unit time

## LIST OF VARIABLES

| | |
|---|---|
| a | Air entry value [m] |
| $a_k$ | Dimensionless exponent [-] |
| b | Campell exponent [-] |
| $b_k$ | Hydraulic conductivity (conductance) under saturation [m $s^{-1}$] |
| +B | Biochemical water production [l $t^{-1}$ MSW] |
| -B | Biochemical water consumption [l $t^{-1}$ MSW] |
| c | Effluent solute concentration [mg $l^{-1}$] |
| $c_i$ | Inflow solute concentration [mg $l^{-1}$] |
| $c_o$ | Initial solute concentration of the pore water [mg $l^{-1}$] |
| D | Dispersion coefficient [$m^2$ $s^{-1}$] |
| ET | Actual evapotranspiration [mm] |
| $ET_o$ | Reference evapotranspiration (referred to grass of 12 cm height during the growing season) [mm] |
| $ET_p$ | Potential evapotranspiration (referred to considered crop) [mm] |
| FC | Field capacity [$m^3$ $m^{-3}$] |
| G | Vapor in gas [l $t^{-1}$ MSW] |
| h | Suction (pressure) head [m] |
| $\Delta H$ | Difference in the hydraulic head [m] |
| K | Hydraulic conductivity [m $s^{-1}$] |
| $K_k$ | Hydraulic conductivity [m $s^{-1}$] at the water content of $\theta_k$ |
| $K_s$ | Saturated hydraulic conductivity [m $s^{-1}$] at the water content of $\theta_s$ |
| $K^A{}_h$ | Horizontal component of the anisotropy tensor of the hydraulic conductivity [-] |
| $K_c$ | Crop coefficient [-] |
| $\overline{K}(\theta)$ | Tensor of the unsaturated hydraulic conductivity [m $s^{-1}$] |
| L | Leachate amount [mm] |
| l | Pore connectivity parameter [-] |
| m | Cumulative discharged solute mass [g] |
| $m_o$ | Initial solute mass of the pore water [g] |
| $m_w$ | Initial mass water content (referred to wet mass) [$dm^3$ $kg^{-1}$] |
| n | Porosity [$m^3$ $m^{-3}$] |
| $n_g$ | Form coefficient of the water retention function [-] |

| | |
|---|---|
| P | Precipitation [mm] |
| p | Pore coefficient [-] |
| $\nabla p$ | Pressure gradient [kg $m^{-2}$ $s^{-2}$] |
| Q | Volume flux [l $s^{-1}$] |
| q | Water flux density [m $s^{-1}$] |
| r | Radius [m] |
| R | Water added (leachate recirculation) [l $t^{-1}$ MSW] |
| S | Water storage inside the landfill [l $t^{-1}$ MSW] |
| $S_e$ | Degree of saturation [-] |
| $\Delta s$ | Length of the media through which water passes [m] |
| t | Time [s] |
| V | Surface runoff [l $t^{-1}$ MSW] |
| $V_{tot}$ | Total volume of the porous media [$m^3$] |
| v | Pore water velocity [m $s^{-1}$] |
| W | Initial mass water content of landfilled MSW [l $t^{-1}$ MSW] |
| z | Coordinate [m] |
| $\alpha$ | Form coefficient in the water retention function [-] [$m^{-1}$], inverse bubbling pressure |
| $\alpha_h$ | Pressure head scaling factor [-] |
| $\eta$ | Dynamic viscosity [kg $m^{-1}$ $s^{-1}$] |
| $\lambda$ | Empirical pore coefficient [-] |
| $\theta$ | Volumetric water content [$m^3$ $m^{-3}$] |
| $\theta_a$ | $\leq \theta_r$ extrapolated volumetric water content [$m^3$ $m^{-3}$] at infinite small matrix potential |
| $\theta_k$ | Volumetric water content [$m^3$ $m^{-3}$] at the hydraulic conductivity $K_k$ |
| $\theta_m$ | $\geq \theta_s$ extrapolated volumetric water content [$m^3$ $m^{-3}$], at full water saturation |
| $\theta_{ma}$ | Water content of macropores participating in the flow process [$m^3$ $m^{-3}$] |
| $\theta_r$ | Residual volumetric water content [$m^3$ $m^{-3}$] |
| $\theta_s$ | Saturated volumetric water content [$m^3$ $m^{-3}$] |
| $\rho$ | Density [kg $m^{-3}$] |
| $\psi$ | Total hydraulic potential [m] |
| $\psi_g$ | Gravitational potential [m] |

| | |
|---|---|
| $\psi_m$ | Matrix potential [m] |
| $\psi_o$ | Osmotic potential [m] |
| $\nabla \psi$ | Gradient of the hydraulic potential [m $m^{-1}$] |

# 1. Introduction

Land disposal of waste has been practiced for centuries. In the past it was generally believed that leachate from waste is purified by soil and groundwater, and hence contamination of groundwater was not an issue (Bagchi, 1990). Thus, disposal of waste in the form of open dumps at all type of sites (e.g., gravel pits, ravines, etc.) was an acceptable practice until the early 20iest century. However, with increasing concern for the environment in the late 1960s landfills become under scrutiny. Within a decade several studies (Andersen & Dornbusch, 1967; Nöring et al., 1968; Zanaoni, 1972; Dunlap, 1976; Kelly, 1976) showed that landfills do significantly contaminate groundwater.

As a result of this finding steps from open dumping of wastes towards sanitary landfills were made. Regulations concerning technical equipment, site characteristics and operation of landfills were enacted and improved with time. Landfill technology has become increasingly sophisticated over the past few decades.

However, in spite of all claimed technical facilities of landfills, gradients of matter and energy between landfill and the surrounding environments still exist. 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 a long term, matter and energy will leave the landfill unless their storage is maintained by a continuous input of energy. Since long term records of the mass flow out of landfills are not available it can 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. The rate of matter leaving the landfill depends on the mass and energy gradient as wells as on the "flow resistance" between landfill and the surroundings, whereby the term "flow resistance" represents physical and chemical barriers, respectively. The aim of modern landfill management is to equilibrate the energy gradient between landfill and the surrounding environment in a controlled manner to a "final storage quality", where the emissions are considered not significantly contribute to natural substance fluxes in soils, air and water (Brunner, 1992). The landfill can become thereby an integrated part of the environment.

Existing landfills of municipal solid waste (MSW) are far from requirements of "final storage quality". Major environmental concerns associated with MSW landfills, containing high content of biodegradable organic matter, are related to the generation of leachate and biogas. Effects of leachate emissions from landfills are local for underlying groundwater and soils

(Ehrig, 1983), whereas production and emission of methane gas posses a global pollution potential since it is a greenhouse gas. It is estimated that solid waste landfills contribute 10 % of the global anthropogenic methane emissions (Watson et al., 1996). The emissions of leachate from MSW landfills will stay on an environmentally incompatible level for hundreds of years (Henseler et al. 1985; Belevi & Bacchini, 1989; Stegmann & Heyer, 1995; Krümpelbeck & Ehrig, 2000). Quantity and quality of leachate and biogas formed depend upon the characteristics of the waste, the design and operation of the landfill and the climatic conditions (temperature, precipitation, and evapotranspiration). In order to stabilize a landfill in a controlled and efficient way, so that environmental impacts are minimized from a short and long viewpoint, understanding of the processes in the landfill interior is crucial. In the last three decades water and water flow were identified as the main factors determining the metabolism of landfills (e.g. Pohland, 1975; Leckie et al., 1979; Bookter & Ham, 1982). Water is on the one hand essential for the biochemical decomposition of organic substances and on the other hand needed for leaching of soluble compounds. Different investigations (Klink & Ham, 1982; Bogner & Spokas, 1993; Christensen et al., 1996) showed that enhanced water flow through waste leads to an acceleration of biochemical processes, as water is the only carrier of substances within a landfill and only water flow facilitates the redistribution of chemicals, micro-organisms and nutrients. Water is also needed for hydrolysis which is the first steep in the anaerobic degradation process.

As water plays the key role in the metabolism, knowledge of water distribution and movement is fundamental for understanding the reactor MSW landfill. Several researchers have pointed out, that in order to improve existing models for describing the landfill behavior, further research must focus on the presence and flux of water (e.g. Straub & Lynch, 1982a; Ehrig, 1983; Augenstein & Pacey, 1991, El-Fadel et al., 1997). A better understanding of water movement inside the waste mass will benefit both the prediction of long term aftercare measurements of existing landfills and the design of strategies for accelerated stabilization, so that burdens on the future generations may be minimized (Beaven et al., 2001).

# 2. Objectives and Scope of Study

The main objective of the presented thesis is to investigate mechanisms governing water flow in MSW landfills. Based on existing approaches and conceptual considerations a mathematical model for describing transport and storage processes of water will be designed.
The development and application of this mathematical model shall enable better insights into the hydraulic behavior of MSW landfills. The calibrated flow model will help to quantify transport processes and make the flow regime in different landfills comparable.
In order to ensure that governing aspects of water movement at field scale are accounted for, the model calibration and validation is carried out using data from full size landfills.

In particular the work addresses the following questions:

- Which model concepts for describing transport and storage of water in MSW landfills have been developed so far?
- What are their benefits, drawbacks and limitations?
- What are the governing mechanisms determining water transport in MSW landfill, and how far are they included in present models?
- Is water flow in landfills comparable to those in soils (as many landfill models so far are adopted from framework carried out for soils)?
- How can leachate generation and its underlying mechanisms be described using existing mathematical formulations?
- To which extent is it possible to reproduce observed leachate generation rates from full scale landfills using a mathematical model?
- Which model parameters determine the heterogeneity of the flow field?
- How big is the fraction of waste mass participating in water flow?

The present thesis can be considered as a framework for the hydraulic analysis of MSW landfills and its description in a mathematical way.

# 3. Landfill Modeling - State of the Art

The evaluation of potential environmental pollution resulting from MSW landfills requires basic knowledge on the inter-relationship between waste materials, landfill technology, operation strategy, biochemical decompositions processes, solute transport mechanisms and precipitation processes. In other words: the governing parameters of the water and solute household of a landfill must be identified. The use of terms like "inter-relationship" and "governing parameters" is already based on an abstraction of the reality into a model. In the background of nearly all scientific considerations models are playing, albeit often unconsciously, an important role.

Models represent reproductions of chosen parts of the reality into artificial systems, so that the fundamental relations are largely held up (Atherton & Borne, 1992).

According to the form of reproduction, models are divided into:

- Physical models (models in the literal sense)
- Mathematical models

Physical models are scaled (usually smaller) and simplified reproductions of the reality, whereas mathematical models use formal descriptions (chemical, physical, empirical or statistical equations) to map the reality or the artificial system, respectively.

In the field of landfill modeling both physical models so called "Landfill Simulation Reactors" LSR (e.g. Stegmann & Heyer, 1995) and different mathematical models are used. Recently applied approaches for predicting long term processes occurring in MSW landfills make use of natural analogous (Bozkurt et al., 2001; Döberl, 2004). This method provides only qualitative results.

The capability of material models (LSR) for predicting the future emission behavior of landfills is limited, because ongoing reactions and processes can only be accelerated to a certain extent. The prevalent method to accelerate physical and biochemical processes going on in the reactors is to enhance the exchange rate of water. The yielded limited time-lapse effect however, is associated with a deliberate modification of prevailing conditions in landfills. To what extent such changes do accelerate only decomposition processes can hardly be quantified. It is generally assumed that enhanced exchange rates of water are walking along with a displacement of emission paths (Scheelhase, 1998).

The great strength of mathematical models is that slow processes ongoing over long periods can be simulated within short time. Furthermore, the ability to predict and evaluate a variety of different scenarios without the effort and expense of physical experimentation is a main advantage of these models. However, it is crucial to remember that mathematical models are idealized representations of physical processes and as such they are driven by assumptions and available input data. In order to validate and assess a model's predictive capabilities it is necessary to verify the model results through comparison with field studies.

The following section reviews in the literature reported mathematical landfill models for simulating the generation of leachate.

## *3.1. Overview of landfill models regarding leachate genesis*

Landfill models can be subdivided according to the matter modeled as follows:

- Water flow models (water balances)
- Solute transport models

Water flow models are designed to conduct water routing and determine the total amount of leachate generated. Solute transport models however, are designed to simulate leachate composition as well. Any simulation of the leachate quality requires information on the generated leachate amount. Thus, solute transport models represent an extension of water flow models.

In the past decades most mathematical models only focused on water balance considerations. The purpose of such studies was to estimate the leachate amount generated within a certain period in order to design necessary storage tanks and treatment plants at the landfill site. Recently major interest has emerged for providing better insights into the reactor landfill. Thus, many model concepts dealing with water flow have been developed, whereas solute transport approaches for landfills trying to predict leachate quality are rarely reported in the literature. This may be partly due to the complex biological, chemical and physical processes involved in landfills that make a mathematical description difficult. The quality of solute transport models, even if they are including the governing biochemical, and physical reactions, is mainly dependent on an adequate reproduction of the water flow processes.

### 3.1.1. Leachate generation models

Leachate generation models are developed and applied to predict water discharge and storage behavior of landfills as well as its migration characteristics inside the waste mass.

According to Ramke (1991) models can be divided into

- *Layer models*
- *Statistical models and*
- *Balance models*

Not included in this classification are those models that are based on the continuum approach (Bear, 1972) of a porous medium. The continuum description assumes that the boundaries between the solid, liquid and gaseous phase of a porous media can be ignored and the physical property in any phase can be described at every point. In the following models based on this concept are summarized as *continuum approach* or *potential models*.

#### 3.1.1.1. Layer models

The concept of the layer model represents the oldest mathematical reproduction of water movement in landfills. The waste body is assumed to be homogeneous and is divided into several horizontal layers. The migration of water is gradually computed from layer to layer, whereby water drainage to underlying layers occurs only when the water content exceeds field capacity FC (water amount which can be held by a porous media against gravity force). This type of water movement is known as the main wetting front.

Remson et al. (1968) were the first who introduced a water flow model for landfills based on the layer concept. The water input for the first layer is assumed to be the difference between precipitation and potential evapotranspiration, whereby the calculations where carried out on a monthly basis. If the water content of the first layer exceeds its storage capacity (field capacity FC), the excess water percolates to the layer beneath. Thus, the landfill body uniformly wets with water from the top to the bottom. Leachate is not generated until the water content in the bottom layer reaches field capacity. Water withdraw from the landfill by evapotranspiration is only considered from the top layer. An upward water movement in the other waste layers due to capillary forces is neglected. The amount of leachate generated at the landfill bottom equals the difference between precipitation and potential evapotranspiration. Due to the use of the potential evapotranspiration the model concept of Remson et al. is suitable for recultivated landfills.

Fenn et al. (1975) improved the above model concept by replacing the potential evapotranspiration through the actual evapotranspiration. Additionally surface runoff was considered. The basic principle of the water flow modeling was kept unaltered. The main advantage of both models for the user was that only few input parameters (field capacity, initial water content) that describe the waste characteristics are required.

Helmer (1974) also predicted leachate generation from landfills using the main-wetting front approach. He modified the model of Remson et al. (1968) by distinguishing between flow conditions at field capacity and below field capacity. At field capcity the water movement is calculated according to Darcy (1856). Below field capacity however, the water flow is controlled by filling the reservoir of each layer. Enhancements of this first concept (Helmer, 1977) include a so called "base leaching" which represents a water discharge even if the water content is below field capacity. The rate of discharge depends on the actual water content of the considered layer. By means of "base leaching" heterogeneities inside the landfill as well as local water saturation should be regarded. The computations were carried out on a daily basis.

At the same time like Helmer Franzius (1977) developed a complex layer model to simulate the water flow in landfills. The starting points for his investigations were experiments in the laboratory using small cells filled with solid waste. He determined the influence of the emplacement density on the hydraulic and hydrologic characteristics of the waste material (field capacity, hydraulic conductivity, infiltration rate and actual evapotranspiration). The attained findings of these experiments concerning relations of different parameters were incorporated in his model approach. Franzius assumed that leachate is not generated until the whole landfill body reaches field capacity. The water flow at field capacity is calculated according to Darcy (1856) using the hydraulic conductivities determined in laboratory experiments. The introduced model is applicable for landfills that are under operation and landfills after closure. Franzius made the following assumptions for his model concept:

- homogeneous landfill body
- uniform water distribution
- constant water storage capacity

Although these assumptions are incorrect for landfills Franzius (1977) never discussed this issue.

The most common model for estimating leachate generation from landfills HELP is also based on the layer concept. The first version of the software code HELP (**H**ydrologic **E**valuation of the **L**andfill **P**erformance) was introduced by Schroeder et al. (1984a, b) at the U.S. Army Engineering Waterways Experiment Station. Up till now several further versions of the original code have been evolved. However, changes of the model concern the implementation of various capping systems or the adjustment to certain climatic conditions only, the basic flow equations stayed unaltered (Schroeder et al., 1994; Berger, 1998).

The original code enables the calculation of:

- surface runoff from the landfill cover according to the Soil Conservation Service (SCS)-curve number method (1973)
- actual evapotranspiration after the modified Penman equation (Ritchie, 1972),
- vertical water movement under saturated and unsaturated conditions and
- leachate discharge at the base sealing of the landfill

The vertical water flow is calculated using a modified form of the Darcy equation (1856), assuming that the hydraulic conductivity is directly proportional to the water content in the single layers. In case that the water content drops below the field capacity the hydraulic conductivity becomes zero. The input parameters required for the model are: precipitation, temperature, solar radiation, humidity at the landfill site, as well as saturated hydraulic conductivity, porosity, field capacity, wilting point of the landfilled waste and the cover layers, respectively. Nowadays after recognizing the complex hydraulic system landfill body and its insufficient reproduction by layer models, HELP is mainly used to estimate the water balance of landfill covers (Ramke, 1991; Berger, 1998).

The main advantages of layer models are the limited number of parameters describing the waste material as well as their easy determinability, and the simple mathematical formulation together with the attempt to incorporate the layer structure of the landfill into the model (Hartmann, 2000). The application of these models enables the operator to estimate the water balance parameters of landfills. However, the capability of layer models for predicting the temporal discharge of leachate must be questioned due to the insufficient assumption of a homogeneous landfill with uniform hydraulic characteristics (conductivity and storativity). Varying storage capacity as well as preferential flow paths are ignored which results (compared to observation at landfills) in overestimating the time for water to discharge from the landfill. Bengtsson et al. (1994) determined by means of the layer model concept that a duration of 10 years is required for the bottom layer of a landfill of 10 m height to reach field capacity and therefore generate leachate. Observations at landfill sites indicate shorter periods. Leachate is already generated short time after waste is landfilled.

#### 3.1.1.2. Statistical and empirical models

Statistical models build or make use of relations between input and output parameters of the investigated system, thereby neglecting internal processes.

One of the first statistical approaches in the field of landfill modeling was introduced by Ehrig (1978). Observed data of weekly precipitation, leachate discharge and potential evapotranspiration of four different landfills over a period of one year were used. Ehrig performed regression analysis using these three parameters. He found a close match between predicted and observed leachate discharge assuming a linear dependency between the actual leachate discharge and the precipitation in the week before. The potential evapotranspiration had to be neglected. Instead of this parameter a seasonal variable was introduced to represent evapotranspiration and storage processes inside the landfill body.

Based on the study of Ehrig (1978) Ossig & Tybus (1986) analyzed precipitation, evapotranspiration and leachate data from eleven landfill sites. The temporal resolution of the observed data varied from days to weeks. The regression analyses based on a daily basis resulted in less agreement compared to analyses carried out on a weekly basis. Ossig & Tybus did not discuss the reason for this result. However, it was shown that water retention characters of different landfills varied significantly (represented by diverse relations between precipitation and leachate discharge).

Jourdan (1981) applied the unit-hydrograph method (Dooge, 1959), which is commonly used in the field of hydrology, to ascertain a correlation between precipitation and leachate generation. Based on single rainfall events and thereby induced discharges the so called unit-hydrograph was determined. A unit-hydrograph gives for a considered hydraulic system the relationship between a single rainfall event of certain duration and the induced discharge. Jourdan neglected in his investigations the base flow of leachate (unaffected by precipitation input), as his aim was to describe extreme precipitation events in order to design drainage and storage facilities for the generated leachate. The application of the determined unit-hydrograph is limited to the investigated landfill and the conditions present during its analysis (cover layers, waste amount, ...). Another drawback beside the limited application is the fact, that the current situation regarding the water storage does not have any influence on the results.

A main disadvantage of statistical models is that they are limited to the specific experiment for which they have been developed. They cannot be extrapolated to simulate other field conditions. Additionally no information about impacts of the landfill body or the operation strategy on the discharge characteristics can be gained. Using this modeling concept it is impossible to get a better insight into leachate formation mechanisms.

#### 3.1.1.3. Balance models

Balance models consider input, output and storage of water in the hydrologic system landfill.

First water balances for landfilled waste (e.g. Quasim & Burchinal, 1970; Fungaroli & Steiner, 1971) were carried out at laboratory columns, where boundary conditions are exactly definable. Water was added by irrigation and evaporation from the waste was prevented by covering the columns. The amount of water stored inside the waste was either determined by weighing the whole column or by sampling and analyzing the waste at the end of the experiment.

Spillman and Collins (1986) reported water balance calculations for landfilled MSW, that incorporate evaporation respectively evapotranspiration. Their concept was based on a linkage between generated leachate and climatic water balance. Data base for their investigations represent measurements carried out at waste lysimeters over a period of five years. Leachate discharge was calculated performing the difference between precipitation and actual evapotranspiration, whereby the actual evapotranspiration was obtained using the potential evapotranspiration and an estimated maximum water storage capacity within the upper waste layers. The water balances were carried out on a weekly basis.

Baccini et al. (1987) conducted element and water balances of municipal solid waste landfills. Unlike conventional investigations considering precipitation, evapotranspiration and leachate only, Baccini et al. accounted for water storage and water input caused by the water content of the landfilled waste. Other parameters like water production and water consumption due to biological degradation processes were identified to be negligibly small. The amount of water stored in the landfill was determined measuring the water content of the landill at undisturbed drilling cores. These measurements indicate that the water storage inside the landfill stays constant over longer periods (years) and equals the initial water content of the landfilled waste material. Baccini et al. applied their balance concept to four landfill compartments of different age. In addition to the water balance considerations material balances were made up for 12 elements (C, N, P, S, Cl, F, Fe, Zn, Pb, Cd, Hg, Cu) resulting in so called transfer coefficients for each element. The transfer coefficient gives the partitioning of the element flux into the gas and liquid phases, respectively.

Water balance models are appropriate to predict the cumulative amount of leachate generated over longer periods. In case of known leachate discharge the concept enables to estimate the amount of water stored inside the waste mass. However, impacts of single rainfall events on the leachate generation cannot be evaluated. Analogous to statistical models hydraulic characteristics of the landfilled waste are not included in this model concept. Nevertheless balance models represent sophisticated tools to identify the governing in- and output parameters of investigated systems.

#### 3.1.1.4. Continuum approach models (potential models)

The continuum approach or potential models are based on the theory of saturated and unsaturated water flow in porous media. They originate from the field of soil science. Assuming that water flow in landfills resembles the water movement in soils, the potential concept was increasingly applied for water flow simulations in landfills. The concept postulates that every movement of water is caused by differences in the hydraulic potential, whereby the total potential $\psi$ of water present at a certain location consists of the matrix potential $\psi_m$ (caused by adsorption and capillary forces) and gravity potential $\psi_g$. Under fully saturated conditions water flow is computed according to Darcy (1856), whereas for unsaturated conditions Richards equation (1931) is applied. Both equations require information about the hydraulic characteristics of the considered porous media (e.g. saturated/unsaturated hydraulic conductivity, water retention characteristics). A general description of the flow equations is given in section 3.2.2.

Reported continuum approach models for describing water tranport in landfills are often combined with solute transport models, in order to predict both, leachate quantity and quality. "Potential" models dealing exclusively with water flow represent exceptions.

The first generation of water and solute transport models considered the landfill body as a homogeneous isotropic media. The reported models differ in the incorporated boundary conditions, in the considered decomposition processes and the mathematical reproduction as well as in the applied solution algorithm for the governing equations. Since the main flow formula is a partial differential equation of 2$^{nd}$ order for which an analytical solution exists only for certain boundary conditions, numerical methods (Method of Finite elements or finite differences) are usually applied to solve this equation.

The first landfill model based on the continuum approach was introduced by Straub & Lynch (1982a, 1982b). The model enabled the operator to simulate water flow and solute transport, dissolving and decay of contaminants in unsaturated sanitary landfills. Water flow through the waste mass was calculated according to Richards equation (1931). The migration of dissolved contaminants is due to convective, dispersive and diffusive transport phenomena. Basic equation for computing these transport processes is the convection-dispersion-equation CDE (Lapidus & Amundson, 1952). Decomposition and dissolving processes of solid waste matters were simulated using first order kinetics. The developed model was applied to simulate data from lysimeter studies. The reported results indicated a fairly good agreement between predicted and observed data, when adjusting the input parameters.

Korfiatis et al. (1984) predicted water flow through refuse of laboratory columns using the theory of unsaturated flow through porous media. The calculations were carried out using Richards equation (1931). Based on the results of their investigations they concluded that little contribution could be attributed to capillary diffusivities when the water content exceeds field capacity. Conversely, diffusion will dominate at water contents below field capacity. Within the scope of model verification the hydraulic characteristics of the waste mass were determined in laboratory tests. The introduced model enabled to simulate the general dynamics of leachate discharge from small scale waste columns.

Based on the investigations of Straub & Lynch (1982a, 1982b) and Korfiatis et al. (1984), Demetracopoulos et al. (1986) developed a mathematical model which incorporates water and solute transport processes in MSW landfills. Compared to the previous studies the numerical techniques for simulating leachate generation and contaminant transport were improved. Another modification of the model concerned the mathematical description of the biochemical decomposition processes. Monod kinetics (1949) was used to characterize these processes. The model enabled to reproduce of the general shape for concentration history in lysimeter studies. The concentration values were shown to be sensitive to microbiological kinetic parameters. Model calibration was difficult due to the limited data on kinetic parameters and virtually none on mass transfer from the solid to the liquid phase. No comparison with actual field data was presented.

Two similar models that operate with the same basic equations were introduced by Lee et al. (1991) and Lu & Bai (1991a, 1991b). Additionally (to the model of Demetracopoulos et al., 1986) boundary conditions for the water flow module like surface runoff and evapotranspiration were taken into consideration. Leachate quality was simply expressed as concentrations of chemical oxygen demand COD. The models were used to simulate data from lysimeter experiments. One difficulty of the model application was the need for many input parameters that are usually not readily available. Indeed a sensitivity analysis showed that at least eight parameters strongly effected the model simulations. Parameters, for which the best simulation results were reportedly obtained, were physically unrealistic.

Vincent et al. (1991) presented a model to describe leachate generation, contaminant transport and biodegradation in landfills. Analogous to the above models, the leachate migration was simulated as an unsaturated flow in a homogeneous medium applying Richards equation (1931). First order kinetics was used to represent dissolving and decay of organic materials. Anaerobic degradation was reproduced by the Herbert kinetics model (a modified form of the Monod model). Leachate quality was evaluated in terms of total organic carbon content TOC only. The model was used to simulate experimental data obtained from waste columns.

Noble & Arnold (1991) developed a one-dimensional finite difference model (FULLFILL) to evaluate water transport and distribution in landfills. The theory of unsaturated flow in porous media built the basis for their model concept. Although the model incorporates biodegradation kinetics to assess leachate characteristics, the emphasis is on water movement within the waste layers. Experiments were conducted in conjunction with the modeling effort to obtain calibration data. The simulation results indicated a fairly good fit with observed data. However, the application was limited to small scale experiments.

Ahmed (1992) presented the two-dimensional unsteady state flow model FILL (Flow Investigation for Landfill Leachate) to simulate the leachate transport in landfills. The model solves the unsaturated-saturated flow equations in a two-dimensional flow field, incorporating surface runoff, and applies the Philip's equation (1969) to compute infiltration rates. The model was applied to simulate the leachate generation in a section of a landfill. Although a good fit with field data was reported, the application of the model is limited to quantify the total amount of leachate generated. Khanbilvardi et al. (1995) compared results obtained by FILL with the landfill model HELP (Schroeder et al., 1984a, 1984b). FILL reportedly indicated a lower value of leachate outflow compared to values obtained by HELP. Although FILL may better represent the field conditions, it is not clear which model provides better estimates because of the uncertainties in its parameters. Leachate quality was not addressed in FILL.

Al-Soufi (1992) introduced a three-dimensional model for simulating water flow and solute transport in soils and applied his concept to landfills. The model addressed saturated and unsaturated water flow, evapotranspiration, plant interception and overland flow from the landfill surface. Chemical and biological processes were represented using simplified empirical equations. The model was used to simulate data from experimental studies.

Although the introduced concept provides a comprehensive framework to predict leachate behavior in landfills, it suffers from the assumption of a homogeneous water distribution and the need of many input parameters.

Al-Yousfi (1992) developed the PITTLEACH model to predict the leachate quantity and quality, together with biogas generation at sanitary landfills (cited in Al-Yousfi & Poland, 1998). Analogous to other models leachate migration was described as unsaturated flow in a homogeneous porous media applying Richards equation. The infiltrated amount of water at the landfill surface was estimated performing a hydrological balance. Anaerobic decomposition of organic waste matter followed three sequential biochemical reactions (hydrolysis, acidogenesis, and methanogenesis). The model was used to simulate several experimental studies. Although it is one of the few modeling efforts that combined gas and leachate generation in one model, leachate and gas transport were modeled independently rather than as a coupled two phase problem.

With the main objective of modeling the landfill gas production Lee et al. (1993) developed the so called LEAGA-I model. LEAGA-I is a one-dimensional model that incorporates a flow module (for unsaturated vertical flow) and a biochemical reaction module, which permits the prediction of methane and carbon dioxide production. Analogous to Al-Soufi (1992) the decomposition processes are subdivided into three sequential reactions (hydrolysis, acidogenesis, and methanogenesis). The model was applied to waste lysimeter tests only.

A concept for a complex landfill model which tries to account for interactions between water movement and gas transport was presented by Swabrick et al. (1995). The intended multiphase flow model should permit the simulation of water, gas and heat transport in sanitary landfills. The model concept is based on the theory of multiphase flow in porous media (Nguyen, 1982). It operates with mass balances for each phase, whereby the single phases are coupled by simple source and sink terms. Water and gas flow through the waste mass were supposed to be calculated according to Darcy (1856). The transport of substances within these two phases should be simulated differently. In the water phase only convective migration is regarded, whereas in the gas phase convective and diffusive transport phenomena are considered. The heat transport should be calculated considering convection and diffusion of heat within the gas and water phase. Additionally to the flow module Swabrick et al. (1995) planned to incorporate a biochemical reaction module that should enable to simulate the anaerobic decomposition of the organic matter. Contrary to other developed models the degradation processes should be included by applying chemical balance reactions. However, the reported concept has not been realized so far into an applicable form.

A complex landfill model including various processes has recently been developed at the Technical University of Braunschweig (Haarstrick et al., 2001; Hanel, 2001; Kindlein et al., 2003). The model incorporates water flow, solute transport, gas movement and heat transport in three dimensions. The main focus however, is on biochemical digestion of the organic matter. In order to study the impact of environmental factors on the biological decomposition processes, pH, temperature and hydrogen changes have been integrated into the degradation model as reaction influencing terms. Water flow in the landfill is calculated using a generalized Darcy law for multiphase flow (Bear, 1972). The waste mass is considered to be homogeneous and isotropic. The movement of substances within the liquid and gaseous phase is attributed to convective and diffusive transport mechanisms. Altogether the introduced model requires 22 input parameters, whereby all of them have been taken from the literature. No calibration or validation using observation data from landfills has been performed yet. Hanel et al. (2001) however, compared model results with data obtained from small scale landfill simulation reactors. They concluded that the general trend of the observed processes could be reproduced quite well applying the mathematical landfill model. In order to reduce the required amount of input parameters, Hosser et al. (2003) developed a method based on the reliability theory which enables to filter the important parameters. Nevertheless the model has not been calibrated and validated at a field scale so far.

All models mentioned above that are based on the continuum approach assume the landfill body to be a homogeneous media, resulting in a uniform water distribution. The assumption of a uniform flow regime is probably valid for laboratory experiments, but not for full scale landfills as tracer experiments show. This postulation is supported by the fact that a good match between observed and predicted (assuming a homogeneous flow field) leachate discharge was only achievable for small scale experiments. However, as different field investigations showed, water distribution in landfills is far from being uniform (Wiemer, 1982; Blight et al., 1992). The water content varies from saturated conditions to complete dryness. This is explained by preferential flow paths that short a large bulk of the landfill. The importance of this phenomenon is undisputed (e.g. Ehrig, 1983; Stegmann & Ehrig, 1989, Zeiss & Major, 1993), but it has been disregarded in most landfill models. Another process that is hardly incorporated in models assuming a homogeneous flow regime is the middle-term storage of water within the landfills. In particular the combination of rapid water transport (immediately after rainfall events) and storage of infiltrated water is impossible to achieve. Indeed, observations at several landfill sites (e.g. Döberl et al., 2002; Åkesson & Nilsson, 1997) prove the importance of these two transport phenomena.

To overcome the limitations mentioned above the landfill body was not considered as a homogeneous media. Attempts were made to divide the landfill into domains with different hydraulic characteristics. The terms of two- or multiply-domain models are used in this context. The consideration of "preferential" flow within the landfill was facilitated.

Within the scope of landfill modeling Young & Davies (1992) were the first who suggested the concept of a two-domain water flow. The flow field (the landfill) was supposed to be divided into a macro- and a micropore-domain with different hydraulic properties. Water flow is calculated separately for each domain applying Richards equation (1931). The main focus of the intended model was given to the degradation processes of organic matter. In particular the generation of landfill gas and its governing factors were described. The model however, did not progress from its initial stage of development. No comparison of the concept with field data is available.

Zeiss & Major (1993) investigated the water flow pattern in small waste columns which were equipped with flow sensors. Even at high water infiltration rates only 28 % of the sensors installed displayed movement of water. The results of their investigations clearly indicate that the water flow is strongly affected by channeling and preferential flow paths, even in small scale experiments. Therefore Zeiss & Major recommended that the waste default values should be revised in developed models, while new leachate generation models need to account for channeled flow (Zeiss & Uguccioni, 1994).
Based on these findings Uguccioni & Zeiss (1997) compared two different approaches to simulate water transport through MSW. They applied the one-dimensional layer model HELP and the two-domain flow model PREFLO (Workman & Skaggs, 1990) for fractured porous media to predict the leachate generation from pilot scale test cells with an average waste volume of 4 $m^3$. PREFLO assumes that the rapid water flow in the channel domain follows Poiseuille's Law (1841) and the lateral water transfer from the channels into the matrix occurs according to Richards' Law (1931). Though PREFLO seems to display the flow processes

physically more realistic than HELP, both models were unable to predict the exact shape of the observed hydrographs. Dependent on the chosen parameter values either the simulated breakthrough time (initial leachate generation) was too long or the cumulative water discharge too high. Due to these unsatisfactory simulation results, Uguccioni & Zeiss (1997) called for a new two-domain model approach that reflects channel and matrix flow.

Bendz (1998) investigated the leachate discharge from landfills and recognized the importance of heterogeneities (particularly fissures and channels) for the water movement. In order to take these heterogeneities into account he introduced a two-domain flow concept composed of channel and matrix domain. Contrary to the approach of Young & Davis (1992) the water flow in the channel domain (macropores) is computed applying the kinematics wave equation according to Beven & Germann (1981). For the matrix domain (represents micropores) Richards equation is used to describe the water movement. The interaction between both flow domains is regarded by simple source and sink terms. In addition to water transport Bendz & Singh (1999) incorporated solute transport processes into the model, whereby only convective movement ("piston flow") is considered. Between the two domains diffusive transport of solutes can take place. Only conservative substances were considered. Tracer experiments in the laboratory were conducted in conjunction with the modeling effort to obtain data sets for the calibration of the model. The concept enabled to simulate the leachate generation and the tracer breakthrough in small scale experiments. An upscaling of the introduced model to real landfill size has not been performed.

Obermann (1999) introduced a one-dimensional mathematical flow model (WATFLOW) to simulate water flow through landfills containing pre-treated waste. Special interest was given to the impacts of the following factors on the water household: landfill geometry, waste pre-treatment and landfill operation. The flow field is again divided into two domains (micro- and macropores). Analogous to Young & Davis (1992), flow in both domains is calculated applying Richards equation. The interaction between the flow domains is different however. According to Obermann water transport within the macropores occurs only if all micropores are fully saturated (threshold concept). The model accounts for varying climatic conditions, increasing landfill height, load dependent settlements and density dependent hydraulic characteristics. Obermann took the variability of the input parameters into account by performing Monte Carlo Simulations (several computations with varying parameters, the result is a possible range of values). WATFLO is limited to water transport. An enhancement of the model concept was presented by Danhamer (2002) who incorporated degradation processes of the organic matter and additionally gas and heat transport.

McCreanor & Reinhart (1996, 2000) applied the model SUTRA (Saturated-Unsaturated flow and **TRA**nsport model) to simulate the water routing in a leachate recirculating landfill. SUTRA (Voss, 1984) was developed to compute water flow and solute transport in two-dimensional variably saturated porous media. The model is based on the Richards equation and the convection-dispersion-equation.

Modeling efforts were done assuming homogeneous anisotropic and heterogeneous waste masses. The heterogeneous waste mass was simulated by applying statistical relationships to the distribution of hydraulic conductivity assigned to regions of the waste mass. The model estimations were verified using data obtained from full scale leachate recirculating landfills. Results from the verification effort indicated that channel flow is the major leachate transport mechanism. However, in order to simulate the leachate generation of MSW landfills in an appropriate way slow Darcian flow need to be considered as well.

Hartmann (2000) evaluated in his thesis the capability of different water flow model concepts (one- and two-domain concepts) to simulate the discharge from bottom ash landfills. Thereto two existing flow models were investigated: HYDRUS-1D (Šimunek, et al., 1998), a conventional potential model (one-domain model) and MACRO (Jarvis, 1991), a two-domain model. Using the two-domain model MACRO Hartmann reported a good fit with field data observed at a bottom ash landfill in Switzerland. Applying HYDRUS-1D however, a good match was only achieved using unrealistic input parameters. The concept of MACRO is based on a partition into macro and micropore-domain. Both domains operate as separate, though interacting, flow regions, each characterized by a degree of saturation, a conductivity and a flux. Richards equation is used to calculate water fluxes in the micropores, whereas the kinematics wave equation according to Beven & Germann (1981) is applied to determine the rapid water flux in the macropores. HYDRUS-1D uses only Richards equation to predict water movement. Contrary to MACRO preferential flow cannot be considered with this model. Hartmann determined the hydraulic characteristics of the bottom ash applying "pedo transfer functions" (PTF) and using information on the grain size distribution. He successfully calibrated and validated MACRO using an annual series of leachate discharge from a bottom ash landfill. However, a close similarity between the water flow pattern in fine grained bottom ash and untreated heterogeneous MSW must be questioned.

The concept of a dual-permeability model seems to be more realistic, though the multitude of required parameters complicates the calibration procedure.

Summarizing the review of reported mathematical landfill models it can be stated that, although several models were reportedly successful to a limited extent in simulating simple cases under well defined and controlled conditions, predicting leachate generation from MSW landfills can still be considered in a developmental stage. The majority of the introduced models neglects the heterogeneous nature of landfills. In rare occasions when heterogeneous flow conditions are considered (two-domain approach) model application was restricted to laboratory tests only. A framework for scaling up the models so that they remain valid on field scale, where the spatial variability must be taken into account, has not been proposed. Furthermore, reported two-domain concepts are invariably derived from the framework that has been carried out for non-uniform water flow in soils. Compared to soils landfills are more heterogeneous, which results in a bigger fraction of preferential flow. This fact was taken into account by adjusting decisive parameters. The underlying assumption however, the similarity between water flow in landfills and soils has not been justified or even discussed yet.

## *3.2. Mathematical description of water flow in porous media*

### 3.2.1. Hydraulic analogues of municipal solid waste landfills

Landfilled MSW is a porous medium with solid material and porespace distributed throughout the volume. The pore space is filled by water and/or gas, whereby the gas can be composed of generated landfill gas and/or intruded air. In general a landfill can be treated as a three phase system, consisting of a solid phase, a liquid phase and a gaseous phase (Figure 3-1).

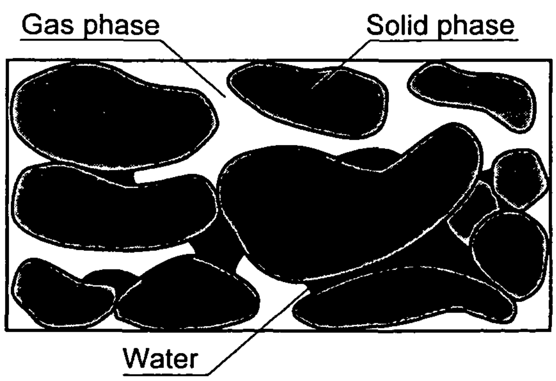



***Figure 3-1*** ***Three phase system (e.g. landfill, soil)***

The porous media which is most comparable to solid waste landfills concerning its structure, porosity, water and gas content, is soil. This finding has already been accounted for in present water flow models for landfills (see chapter 3.1.1), as all introduced concepts so far are derived from the framework which has been carried out for soils. However, in spite of general similarities between both porous media, the composition of landfills is more heterogeneous (e.g. grain size distribution, shape and hydraulic properties of single grains, organic matter) compared to soils. This heterogeneous character of landfills leads to a highly non-uniform distribution of water. Saturated zones and completely dry zones are found next to each other. The water flow in landfills is dominated by preferential pathways that short a large bulk of waste (e.g. Zeiss & Major, 1993).
Recently developed mathematical landfill models, that incorporate preferential flow (two-domain approaches, see chapter 3.1.1.4), try to account for the heightened heterogeneous characteristics of MSW landfills by simply adjusting decisive parameters.

Beside the pedosphere also the lithosphere represents to some extent a comparable medium for describing the hydraulic behavior of sanitary landfills. In particular the phenomenon of preferential flow is partly explained using analogues in the lithosphere. Döberl (2004) compares the hydraulic system landfill with karst formations that show similar discharge characteristics. In karst systems so called karst tubes (representing preferential flow paths) are accountable for the quick response of discharge after precipitation. Whereas the fine-grained dolomite releases water slowly and therefore provides discharge during dry periods ("base-flow").
Figure 3-2 gives an example for the hydrology of karst systems. Despite comparable discharge characteristics of karst formations and MSW landfills, genesis and development of preferential flow in both media is different. In karst systems favored flow paths are formed by dissolution of soluble rocks. In landfills the heterogeneity of the waste disposal is responsible for flow channels.

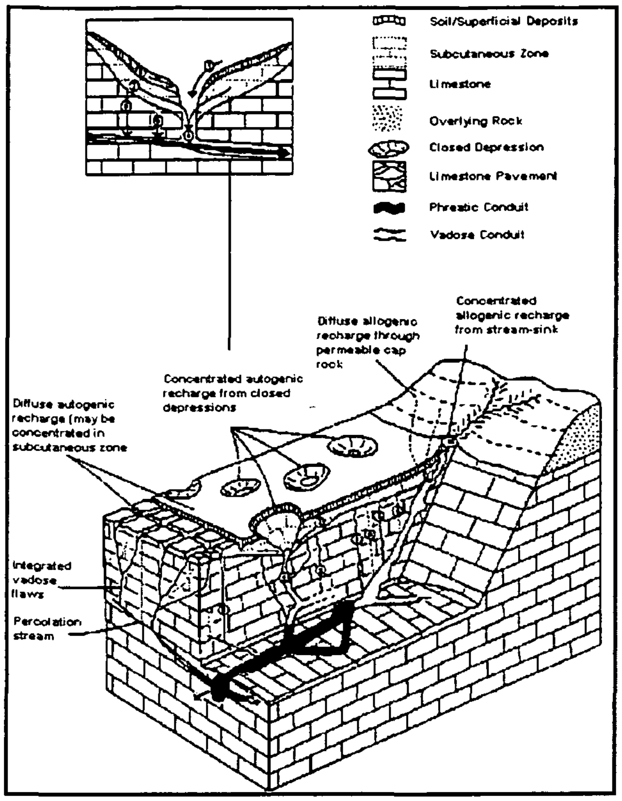



*Figure 3-2* *Hydrology of karst system (Crawford Hydrology Laboratory, www.dyetracing.com)*

Attempts to model the karst hydrology range from simple Darcy flow assumptions with an average rock permeability, over two-domain approaches (rock permeability and conduit permeability) to so called discrete models, that require explicit information about the spatial extent of each conduit (White, 2002). The discrete modeling concept is inapplicable for landfills, as an exploration of single flow paths in landfills is impossible. The other two model approaches (Darcian flow, two-domain flow) correspond to the concepts for simulating water flow in homogeneous respectively fissured soils. These approaches have already been applied for sanitary landfills (see chapter 3.1.1.4).

### 3.2.2. Equations describing the water flow in porous media

In the following a short overview of mathematical equations for describing water flow in porous media is given.

Darcy (1856) made an important discovery for ground water flow. He proved in his experiments a linear relationship between the water flux density through a saturated sand column and the forces acting on the water.

*Darcy equation (one-dimensional flow):*

$$q = -K_s \frac{\Delta H}{\Delta s}$$ **Equation 3-1**

| | |
|---|---|
| $q$ | *Water flux density [m s⁻¹]* |
| $K_s$ | *Saturated hydraulic conductivity [m s⁻¹]* |
| $\Delta H$ | *Difference in the hydraulic head [m]* |
| $\Delta s$ | *Length of the media through which flow passes [m]* |

When the water flow is three-dimensional the equation can be generalized to:

$$q = -\overline{K} \cdot \nabla \psi$$ **Equation 3-2**

| | |
|---|---|
| $\overline{K}$ | *Tensor of the saturated hydraulic conductivity [m $s^{-1}$]* |
| $\nabla \psi$ | *Gradient of the hydraulic potential [m $m^{-1}$]* |

The Darcy equation is applicable for water flow under the condition that all pores are filled with water. Combining Darcy equation and the approach of Buckingham (1907), which assumes that the hydraulic conductivity is function of the water content, with the law of conservation of mass results in *Richards equation* (1931). This equation is applied to calculate the water flow in variably saturated porous media.

$$\frac{\partial \theta}{\partial t} = \nabla\left(\overline{K}(\theta) \cdot \nabla \psi\right)$$ **Equation 3-3**

| | |
|---|---|
| $\theta$ | *Volumetric water content []* |
| $t$ | *Time [s]* |
| $\overline{K}(\theta)$ | *Tensor of the unsaturated hydraulic conductivity [m$s^{-1}$]* |

Richards equation assumes that the unsaturated hydraulic conductivity K(θ) is a function of the water content θ, which is again a function of the matrix potential $\psi_m$. The matrix potential

$\psi_m$ can be understood as a measure for the intensity of water fixation inside a porous media. It represents the capillarity of the single pores. The total hydraulic potential $\psi$ of water inside a porous media is defined as the sum of matrix potential $\psi_m$ and gravitational potential $\psi_g$. In rare occasions also the osmotic potential $\psi_o$ of the water can be of importance (e.g. root water uptake).

$$\psi = \psi_m + \psi_g \qquad \textbf{\textit{Equation 3-4}}$$

The matrix potential ranges from 0 cm, which represents water saturation of the pore space, up to $-10^7$ cm. High absolute values of $\psi_m$ correspond to low water contents. The relationship between water content and matrix potential is known as water retention characteristics. Figure 3-3 gives an example for the retention characteristic curve of a soil.

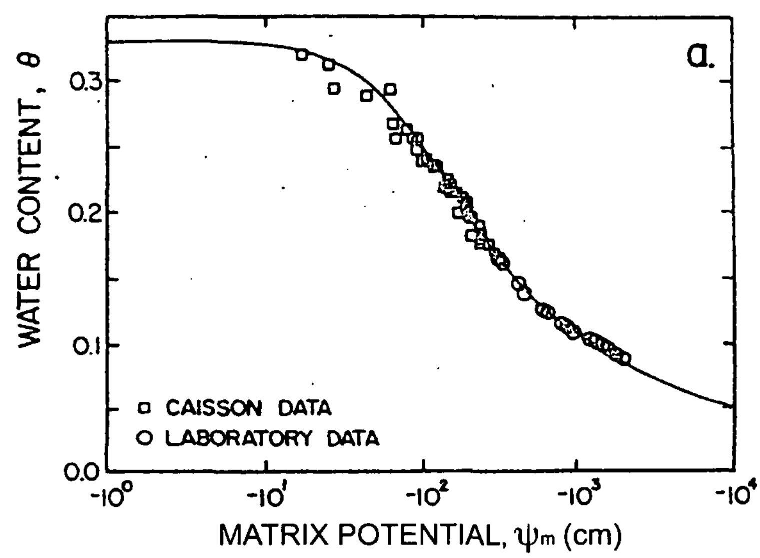



***Figure 3-3*** ***Water retention curve of a soil (van Genuchten & Šimunek, 2002)***

Several empirical equations describing the retention characteristics of soils have been developed (e.g. Brooks & Corey, 1964; Campell, 1974; Hutson & Cass; 1987; van Genuchten, 1980). Examples for these equations are given in appendix 8.2.

The hydraulic conductivity $K(\theta)$ is strongly dependent on the water content. It decreases with decreasing water content. The maximum value is reached at full water saturation. At this point it equals the saturated hydraulic conductivity $K_S$.

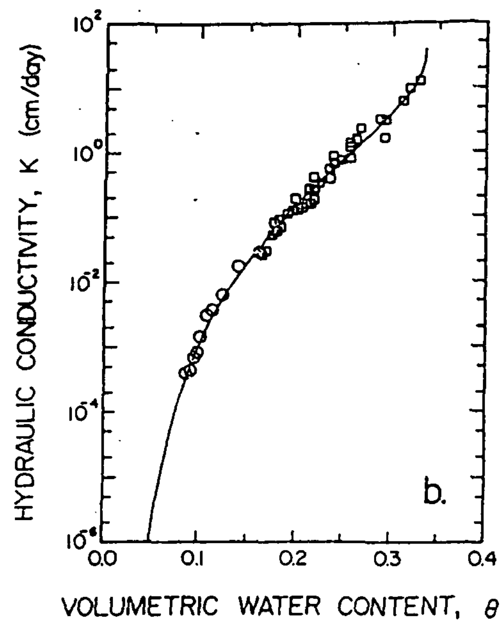



*Figure 3-4 Hydraulic conductivity versus water content (van Genuchten & Šimunek, 2002)*

Both functions K(θ) and ψ(θ) are dependent on the pore size distribution of the considered media.

The application of Richards equations implies that the water flow is determined by both matrix and gravity potential. However, with increasing pore diameter the impact of the porous matrix on the water flow declines. Thus, gravity is the only driving force for water transport through macropores. Another difference between water flow through fine-pored media and macropores is that water moves much faster through larger pores due less flow resistance ("friction forces"). The phenomenon of rapid water flow along "preferential pathways" was first recognized by Schumacher (1864), who stated that "the permeability of a soil during infiltration is mainly controlled by big pores, in which the water is not held under the influence of capillary forces". Several attempts to describe these flow processes in a mathematical way have been made so far. In order to simplify the mathematical description the network of preferential flow paths is usually lumped into one channel. In the following, the most common approaches for modeling preferential flow through soils respectively fractured rocks are briefly presented. All of them have already been applied to simulate the water flow through sanitary landfills.

- Poiseuille Law (1841)

$$Q = \frac{\pi \cdot r^4}{8\eta} \nabla p$$ ***Equation 3-5***

| | |
|---|---|
| $Q$ | *Volume flux [m³ s⁻¹]* |
| $\eta$ | *Dynamic viscosity [kg m⁻¹ s⁻¹]* |
| $r$ | *Radius of the pipe [m]* |
| $\nabla p$ | *Pressure gradient [kg m⁻² s⁻²]* |

This equation was developed to calculate steady state, laminar water flow through cylindrical pipes. Uguccioni & Zeiss (1997) applied this equation (implemented in the model PREFLO) to compute preferential water transport through waste columns.

- Kinematic wave assumption (after Beven & Germann, 1981)

$$q = b(\theta_{ma})^a$$ ***Equation 3-6***

| | |
|---|---|
| $q$ | *Volume flux density [m s⁻¹]* |
| $b$ | *Hydraulic conductivity (conductance) under saturation [m s⁻¹]* |
| $\theta_{ma}$ | *Water content of macropores participating in the flow process [m³ m⁻³]* |
| $a$ | *Dimensionless exponent [-]* |

It is assumed that the water flow through macropores follows a power function of the water content. This approach was used by Bendz (1998) and Hartmann (2000) to simulate the water flow through landfills.

- Richards equation for the macroporic flow

In a so called dual-permeability approach (Gerke & van Genuchten, 1993, 1996) the macropores are assigned a higher hydraulic conductivity than the micropores. The water flow is calculated separately for both domains (macropores and micropores). Thus, for each point of the flow field two velocities, two water contents and two hydraulic heads exist. The landfill model of Obermann (1999) is based on this approach.

Although, the importance of macroporic flow for subsurface hydrology is undisputed, at present the application of complex models taking this phenomenon into account is restricted to theoretical investigations and laboratory studies (Šimunek et al., 2003). In particular the large numbers of parameters involved and the current lack of standard experimental techniques to obtain them, inhibit the employment.

The approaches originating from soil physics can only be adapted to landfills, under the condition that flow mechanism and underlying assumptions are applicable for preferential flow in landfills.

# 4. A New Approach for Modeling Leachate Generation

Due to the limitations and drawbacks of existing landfill models a new approach for predicting leachate generation and water storage in MSW landfills is required. The two main principles for this model are: universal applicability (in particular variable boundary conditions at field scale) and incorporation of heterogeneous water flow. Both principles have not been combined so far in existing mathematical landfill reproductions.

In the following chapters a new approach for modeling water flow in MSW landfills is gradually derived. Starting from simple black box considerations (water balances) the determining factors and processes evolved are identified and discussed.

## *4.1. Water balance considerations*

The simplest way to describe the hydrology of landfills is performing a water balance. Thereby only input, output and storage of water into or out of the system are considered. This general concept of modeling is known as system identification technique and the resulting model is called black-box model (Bender, 1978).

The general water balance equation for MSW landfills can be written as follows (modified after Baccini et al., 1987):

$$P + W + R \pm B - (ET + G + L + V) - S = 0 \qquad \textbf{\textit{Equation 4-1}}$$

Water is introduced into the landfill through the moisture of the landfilled waste material (W), as precipitation (P) and in some landfills by water addition during landfilling and recirculation of leachate (R). Some of the precipitation may run off as overland flow (V), and some may evaporate from the waste material or be removed by transpiration from the vegetation cover (ET). Inside the landfill body some water may be generated or consumed by biochemical processes (B), whereby anaerobic decomposition processes of organic matter consume water and during aerobic decomposition water is produced. The remaining water must accumulate (water storage S) or be discharged by drainage (leachate L). Beside the drainage of leachate water may also leave the landfill as vapor (G) through the gaseous phase. The storage of water against gravity is caused by the texture of the waste material itself (materials with capillarity or water retaining properties) as well as by the texture of the landfill (capillary voids between waste materials).

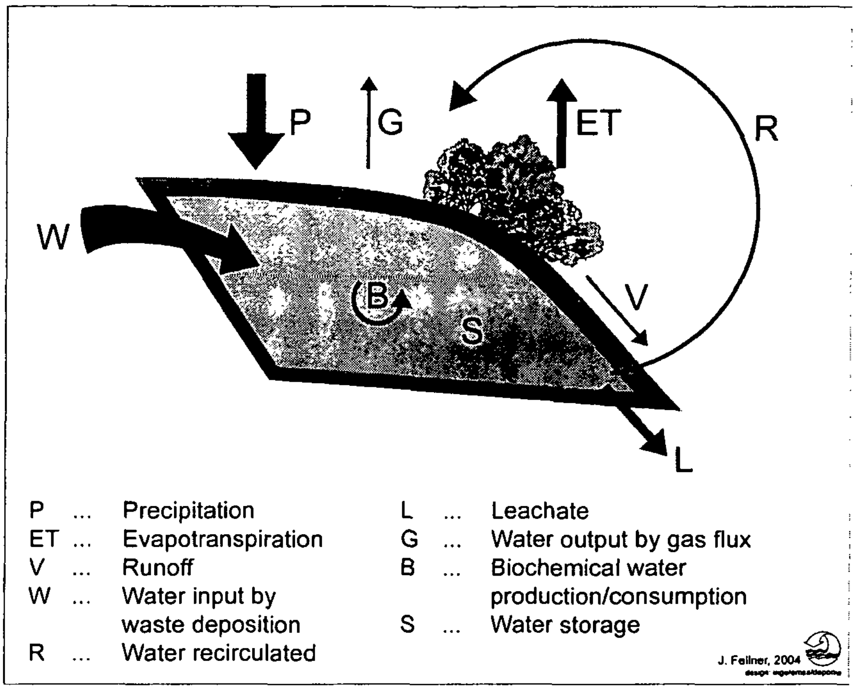



*Figure 4-1* *Water balance of landfills*

To provide a simple example, water balance investigations into the experimental landfill "Breitenau" (description of the landfill site see chapter 5.1) are briefly presented.

The water input into the landfill Breitenau caused by precipitation (P) was evaluated using mean values of rainfall obtained from three meteorological stations (Neunkirchen, Saubersdorf and Wr. Neustadt) nearby the landfill site (see Figure 5-3). Due to the lack of measurements concerning the initial water content of the landfilled waste (W), this parameter was estimated using literature data. According to Brunner et al. (1983) and Ehrig (1989) an initial water content of MSW of 30 % (WS) was assumed. This figure is in agreement with water content measurements of MSW originating from the same community the landfilled waste came from (Schachermayer et al., 1994). The amount of water added during landfilling, re-circulated leachate (R) and surface runoff (V) were calculated according to data given in Riehl-Herwirsch et al. (1995). Water was added during the landfilling of the waste in order to ensure a high water content, and thus, better conditions for decomposing microorganisms. The rate of potential evapotranspiration was evaluated after Haude (1954) using regionally adapted factors of phenology (Dobesch, 1991) and crop coefficients according to the vegetation cover. In order to obtain the required actual evapotranspiration for the water balance calculations the availability of water in the cover layers had to be considered. Thereto the water flow model LEACHW (Hutson & Wagenet, 1992) was applied. This model enables to compute the actual evapotranspiration in dependency of the available water over the root depth. Due to several assumptions concerning evapotranspiration the determined values are highly uncertain. The leachate discharge (L) was measured using different methods during the considered time period (1987 – 2002). Electromagnetic flow meters, mechanical gauges and differences in the water level of the leachate collection tanks were used to determine the leachate generation rate. The amount of condensate in landfill gas (G) was supposed to be 60 g water per $m^3$ gas (Rettenberger, 1987). The biochemical water production/consumption (±B) was computed according to Pöbel (1964) for aerobic degradation processes, and according to Stegmann & Ehrig (1980) for anaerobic processes. The amount of water stored within the landfill body (S) was estimated by solving Equation 4-1.

The "Breitenau" landfill is divided into three separate compartments with different capping systems (see Figure 5-1). Table 4-1 gives the results of the water balance calculations for all three compartments including the uncertainties. The values are referred to the initial moist mass of the landfilled waste in order to obtain comparable data.

***Table 4-1 Parameters of the water balance (1988-2002) for the landfill Breitenau [l/t MSW]***

| | | Compartment I | Compartment II | Compartment III |
|---|---|---|---|---|
| Landfill cover (surface) | Precipitation (P) | 1,230±20 | 1,060±20 | 1,130±20 |
| | Surface run-off (V) | -63±6 | -26±3 | -15±2 |
| | Evapotranspiration (ET) | -785±80 | -750±75 | -965±50 |
| | *Climatic water balance (P-ET-V)* | *382±82* | *284±78* | *150±54* |
| Landfill | Water added (R) | 51±3 | 66±3 | 112±6 |
| | Initial water content (W) | 300±30 | 300±30 | 300±30 |
| | Water production (+B) | 11 | 11 | 11 |
| | Water consumption (-B) | -15 | -15 | -15 |
| | Vapor in gas (G) | -3 | -3 | -3 |
| | Leachate (L) | -356±18 | -246±12 | -113±6 |
| | Storage (S) (rounded) | 370±90 | 400±85 | 440±60 |

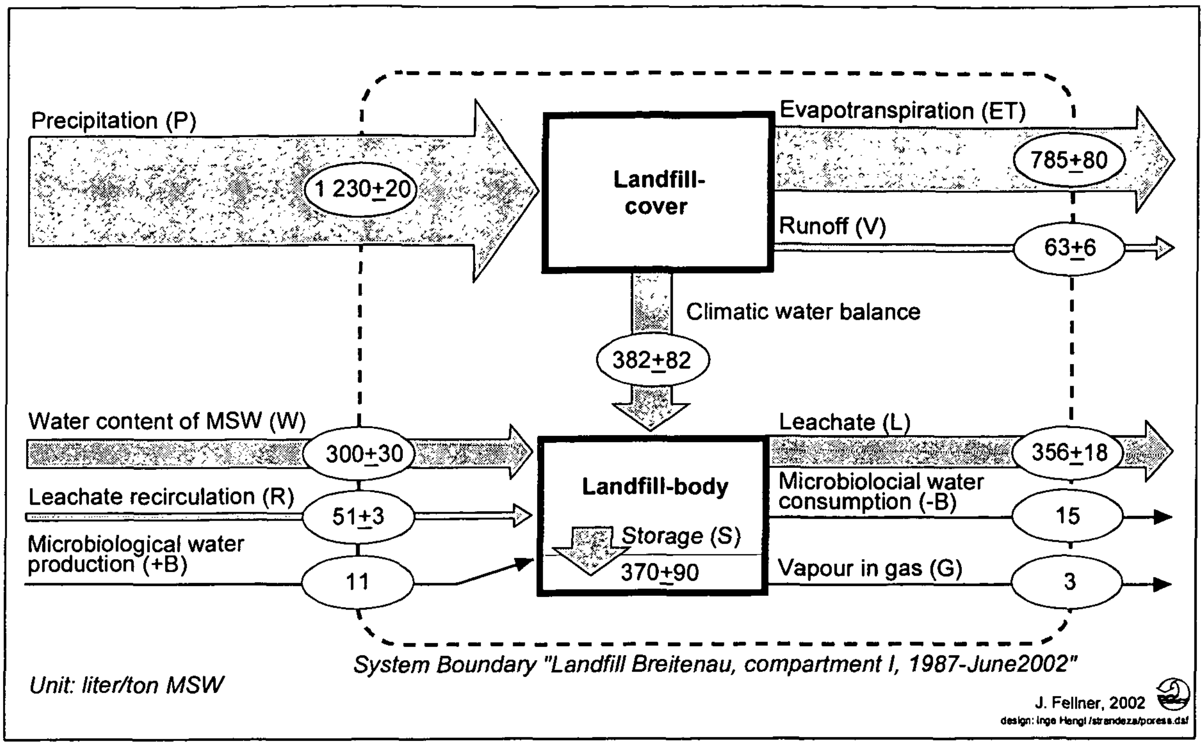



*Figure 4-2 Water balance for the Breitenau landfill – Compartment I*

Precipitation, evapotranspiration and runoff can be summarized to the climatic water balance. This parameter represents the amount of water that percolates from the landfill cover layers into the waste body.

The results indicate that biochemical water production and consumption as well as water losses due to the vapor in landfill gas are non-relevant for the water balance of landfills. Thus, they may be neglected in further considerations. A comparison of the results shows that the water input into the three compartments of the landfill was different during the observation period (15 years). This attributes on the one hand to different capping systems and on the other hand to diverse operation strategies.

Water input expressed as climatic water balance is governed by the capping system and the climatic conditions at the landfill site. This water input turns out to be highest in Compartment I (382 l/t MSW), followed by Compartment II (284 l/t MSW) and Compartment III (150 l/t MSW). Different evapotranspiration is mainly responsible for the diverse water input. The geometry of the compartments (average landfill height) also affects the rate of water coming into contact with waste. Compartment III that is abundantly covered with vegetation shows the highest quantity of evapotranspiration. Plants withdraw more than 85 % of the incoming precipitation.

The second major water input into the landfill beside the climatic water balance is the water content of the landfilled MSW. Furthermore, in the considered case of Breitenau landfill water was added during landfilling, and after landfill closure in the form of re-circulated leachate. For the three compartments differences in the additional water input were reported. At Compartment III recirculation of leachate was carried out till 1995 with a total additional water input 112 l/t MSW. At Compartment I and Compartment II water recirculation was stopped after landfill closure in 1989. Till this date around 51 l/t and 66 l/t MSW have been recirculated at C I and C II, respectively.

Summarizing climatic water balance, irrigation water and initial water content leads to a total water input of around 730 l/t MSW at C I, 650 l/t MSW at C II and 560 l/t MSW at C III. During the same period leachate generation rates at C I of 356 l/t MSW, at C II of 246 l/t MSW and at C III of 113 l/t MSW were registered. This results in an average water storage of around 370, 400 and 440 l/t MSW for the three compartments.

The water balance of the Breitenau landfill is dominated by the following parameters:

- precipitation
- evapotranspiration
- initial water content of the landfilled MSW
- water storage
- leachate
- re-circulated leachate and water added during landfilling
- runoff

Whereby, the relevance of the term re-circulated leachate and water addition during landfilling is specific due to the certain operation strategy at the landfill site Breitenau. Although the importance of other terms is site specific, their general importance can be assumed.

Model approaches for simulating water flow in landfills should incorporate the above mentioned input and output parameters.

Apart from identifying the parameters determining for the water balance of landfills a phenomenon, unaccountable by black-box considerations, can be observed by comparing water entering and exiting the three compartments:

Higher water input into landfills consequentially results in higher leachate discharge, which is not surprising. Higher water input however, does not inevitably increase water storage inside the waste body. For instance, Compartment III with the lowest water input of 560 l/t MSW shows the highest rate of water retention (440 l/t MSW) and Compartment I with the highest water input of 730 l/t MSW shows less storage of water (370 l/t MSW). The transfer coefficients describing the relation between water input, output and storage are significantly diverse for each compartment (see Figure 4-3).

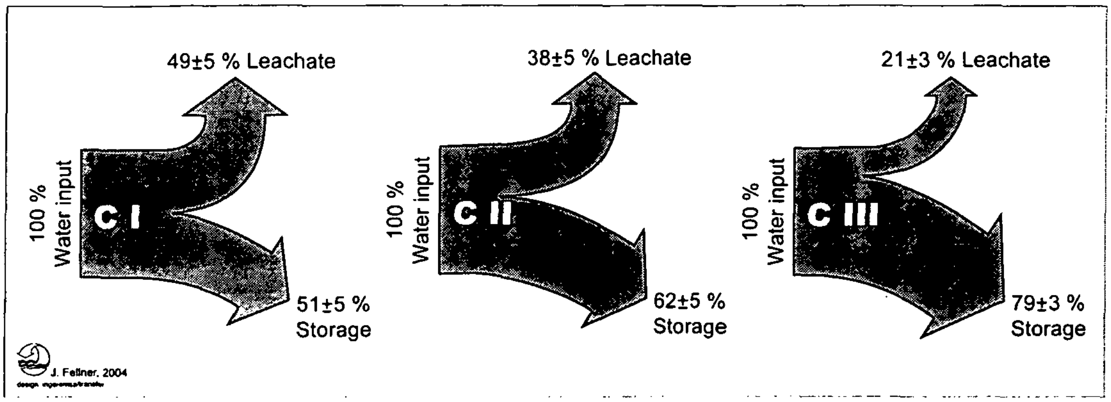



***Figure 4-3 Water transfer-coefficients for each compartment (Breitenau landfill)***

Compartment I shows the highest fraction of water release (49±5 %), whereas in Compartment III only 21±3 % of the water input occurs as leachate discharge at the bottom of the landfill. C II lies with 38±5 % of released water in between.

Transfer coefficients for the different water paths depend on the amount of water input, as the storage capacity of a landfill is limited. In a long term view the transfer coefficient for the leachate path must converge to 1. The different transfer coefficients deduced for the three compartments however, may only partly attribute to various water inputs. The absolute figures also indicate least water storage at Compartment I.

Considering that similar waste was landfilled and water input was within the same range, the transfer coefficients at the three compartments should be in the same range. Since similar hydraulic characteristics (storage capacity) of the waste material itself can be assumed. The results of the water balances however, show different hydraulic behavior for each compartment. This fact indicates that the water flow through landfills is not only dependent on the characteristics of the waste material itself.

Low absolute values together with low transfer coefficients for the water storage indicate a highly non-uniform water distribution in landfills. In this case water flow is funneled to less bulk of waste and therefore also the effective storage capacity is restricted to the waste mass participating in water flow. Applying this postulation to the results of the Breitenau landfill would mean that large zones in C I did not get in contact with water yet. Whereas according to the results of C II and C III it can be assumed that dry zones in these compartments are of less importance. Thus, the water flow regime in Compartment II and C III is more uniform compared to C I.

Although comparisons of simple balance considerations may already provide an indication on water distribution in landfills, it is imperative for understanding and studying leachate generation processes to investigate the hydraulic characteristics of landfilled MSW. Obviously, these properties will have a direct impact on the results of any project studying leachate routing just as the hydrologic properties of the subsurface media will affect a groundwater modeling study.

## *4.2. Hydraulic characteristics of municipal solid waste*

Municipal solid waste is due to its origin a highly heterogeneous media. Investigations of Turczynski (1988) showed that the grain size varies from smaller 0.5 mm up to 1000 mm. The hydraulic behavior of single waste components is divergent. It ranges from highly water adsorbent (e.g. paper, textile) to water repellent, from impermeable (e.g. plastic foils) to well permeable materials. Nevertheless, it is possible to determine hydraulic parameters for landfilled waste, if the investigated volume is big enough to be representative for the mixture of materials. Usually

- hydraulic conductivity
- porosity and
- water retention characteristics

are used to characterize the hydraulics of sanitary landfills. In the present chapter an overview of reported hydraulic parameters is given.

### 4.2.1. Hydraulic conductivity

The hydraulic conductivity K represents a measure of the ability of a substance to transmit water. This parameter determines together with the porosity the velocity of a fluid inside the porous media and is therefore important for hydraulic considerations. Figure 4-4 gives a summary of the saturated hydraulic conductivity of MSW from different investigations. Some of the experiments were carried out for different roof pressure resulting in altered waste densities. The reported conductivity values vary between $1\times10^{-2}$ and $5\times10^{-9}$ m/s. The higher values represent waste with a low degree of compaction and measurements performed in-situ by pumping tests, since they contain a larger horizontal component and the conductivity in the horizontal direction is nearly 10 times higher than in the vertical direction (Ramke, 1991; Powrie & Beaven, 1999). With increasing waste density a significant decrease of the hydraulic conductivity is apparent. The rate of the conductivity decline however, is strongly diverse in different experiments.

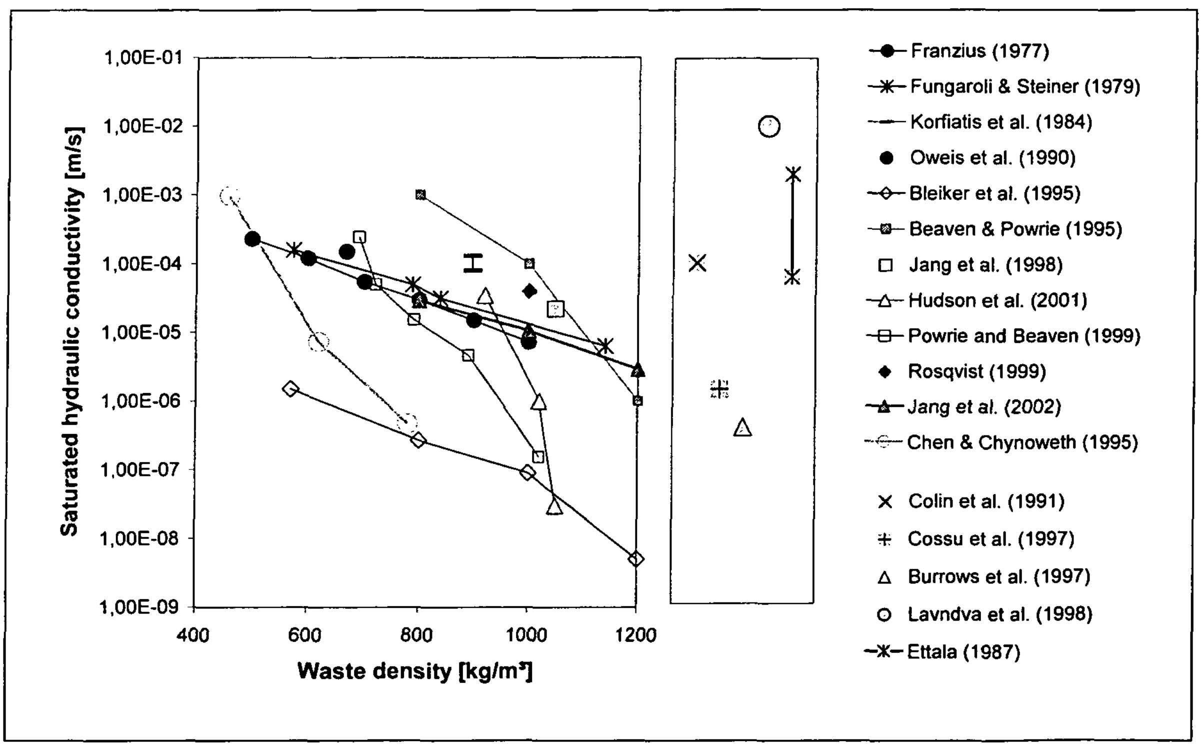



*Figure 4-4 Saturated hydraulic conductivity of MSW*

The saturated hydraulic conductivity $K_s$ represents the ability to transmit water under the condition that all pores are filled with water. Modern sanitary landfills that are operated according to the state of the art (equipped with a leachate collection system at the landfill bottom) are far from water saturation. Thus, water transport within the landfill occurs mainly under unsaturated conditions. It is therefore crucial for water flow investigations to characterize the unsaturated hydraulic conductivity performance of MSW landfills. Hydraulic investigations have primarily been conducted under water saturation. Only a few studies for unsaturated waste were reported in the literature. The results of these studies are summarized in Figure 4-5. Apart from the investigation carried out by Jang et al. (2002) a strong decline of the hydraulic conductivity with increasing matrix potential $\psi_m$ (suction head) is noticed. This means that the permeability of MSW is highly dependent on the water content and the degree of saturation. Low water content is associated with low hydraulic conductivity.

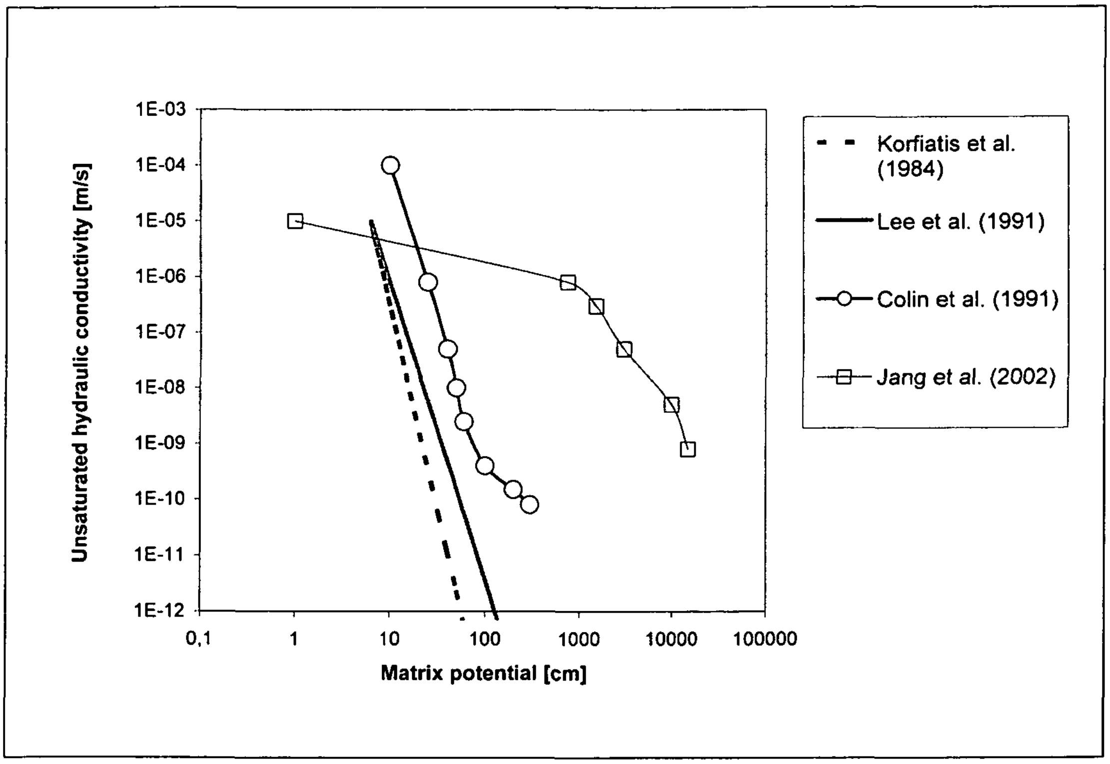



*Figure 4-5 Unsaturated hydraulic conductivity of MSW*

### 4.2.2. Porosity

The porosity n is a measure of the void space in a porous media. It is defined as the ratio between pore volume and total volume. Under saturated conditions the void space is totally occupied by water. The porosity is relevant for hydraulic considerations as it determines the upper limit of the water storage capacity.

In the literature porosity values of MSW landfills between 0.30 and 0.65 were reported (see Table 4-2), whereby the majority of the values lies around 0.50. The porosity of landfilled waste logically decreases with increased compaction energy (Franzius, 1977).

***Table 4-2*** ***Porosity of landfilled MSW reported in the literature***

| Reference | (wet) Density ρ [kg $m^{-3}$] | Porosity n [$m^3$ $m^{-3}$] |
|---|---|---|
| Pacey (1982) | 890 | 0.48 – 0.51 |
| Oweis et al. (1990) | 640 – 1,300 | 0.40 – 0.50 |
| Landva & Clark (1990) | 1,000 – 1,400 | 0.30 – 0.60 |
| Colin et al. (1991) | 400 | 0.65 |
| Zeiss & Major (1993) | 360 – 550 | 0.47 – 0.58 |
| Powrie & Beaven (1999) | 690 – 1,020 | 0.46 – 0.56 |
| Rosqvist (1999) | 1,000 | 0.53 |
| Yuen et al. (2001) | 840 | 0.55 |
| Jang et al. (2002) | 800-1,200 | 0.29 – 0.52 |

### 4.2.3. Water retention characteristics

The water retention characteristics describe the capillarity of a porous media. It is important for the storage of water inside the landfill. The retention characteristics can be expressed using either a single parameter (field capacity) or a defined relationship between water content θ and matrix potential $\psi_m$ (suction head) resulting in the so called water retention curve. The field capacity (FC) gives the maximum amount of water (per volume) that can be retained against gravity force. Typically reported field capacities for MSW landfills range from 0.12 to 0.54 (Table 4-3). Field capacity is basically a function of the waste composition, the density and the porosity. It is expected to change with time, as the degradation of the waste alters its composition (Blight et al., 1992). The time for the water content to increase from its initial value to field capacity can be significant. Bengtsson et al. (1994) found that water was still accumulating in 10-year-old landfills.

*Table 4-3 Field capacity of landfilled MSW reported in the literature (after Yuen et al., 2001)*

| Reference | Field capacity [$m^3\ m^{-3}$] |
|---|---|
| Remson et al. (1968) | 0.29 |
| Franzius (1977) | 0.16 – 0.45 |
| Holmes (1980) | 0.29 – 0.42 |
| Straub & Lynch (1982°) | 0.30 – 0.40 |
| Korfiatis et al. (1984) | 0.20 – 0.30 |
| Canziani & Cossu (1989) | 0.29 – 0.37 |
| Oweis et al. (1990) | 0.20 – 0.35 |
| Colin et al. (1991) | 0.40 |
| Lee et al. (1991) | 0.32 – 0.54 |
| Zeiss & Major (1993) | 0.12 – 0.14 |
| Schroeder et al. (1994) | 0.29 |
| Bengtsson et al. (1994) | 0.44 |
| Powrie and Beaven (1999) | 0.40 – 0.45 |
| Rosqvist (1999) | 0.41 |
| Yuen et al. (2001) | 0.34 |
| Jang et al. (2002) | 0.26 – 0.45 |

A more detailed characterization of the water holding capability of a porous media is the so called water retention curve. This approach conceptually treats the waste as a bundle of capillary tubes with a range of diameters. The distribution of pore diameters is a material characteristic. It determines the water retention curve that represents the relation between water content $\theta$ and matrix potential $\psi_m$ (or suction head). Several empirical equations have been developed so far to describe this relationship for soils (Brooks & Corey, 1964; Mualem, 1976; van Genuchten 1980; Hutson & Cass, 1987; Vogel & Cislerova; 1988). In order to depict the retention characteristics of waste, parameters from soil models have been simply adjusted. Figure 4-6 gives on overview of the reported water retention curves for MSW. The results vary over a range of two magnitudes (in matrix potential). The strong distinctions are mainly due to different waste composition, compaction, observation scale and measurement method. Additionally wetting and drying cycles have an impact on the retention characteristics (Zeiss & Major, 1993). Despite the large variations at least a general shape of the water retention curves for MSW can be derived from Figure 4-6.

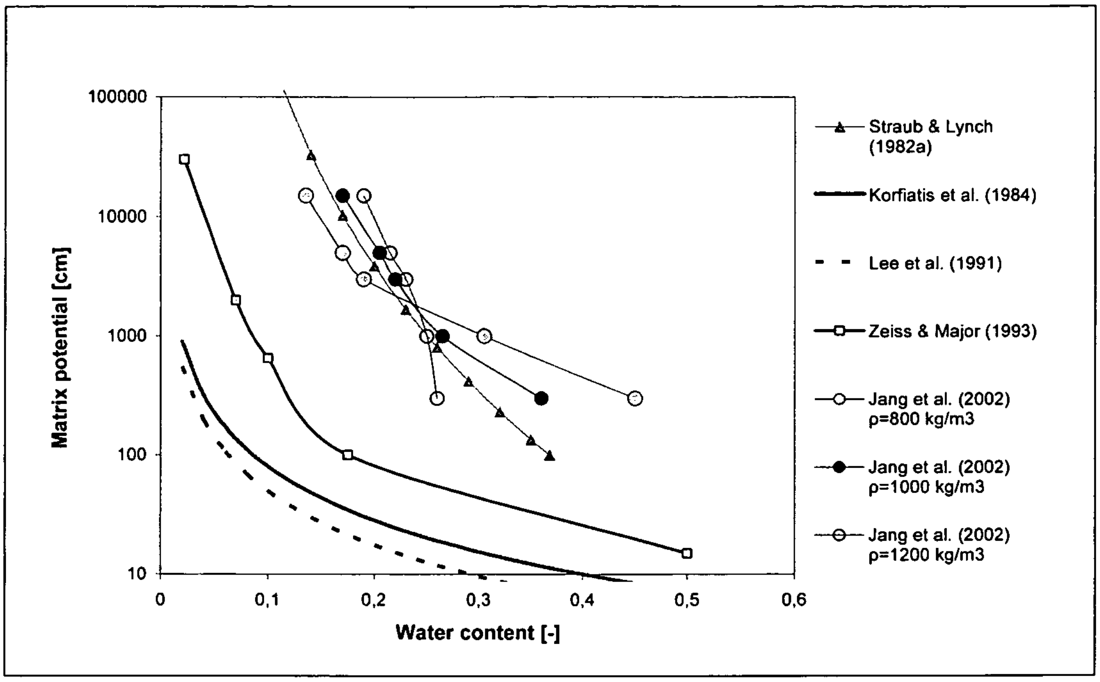



*Figure 4-6 Water retention characteristics of MSW*

All the above mentioned parameters depend on various factors. Waste composition, the degree and way of compaction, the deposition age, the degradation state, the observation scale and also the measurement method govern the hydraulic characteristics of MSW. Therefore, parameters reported in the literature describing the hydraulic properties show large variations. Consequently, a literature review can only give a feasible range of the parameter values.

## *4.3. Leachate hydrographs*

The discharge hydrographs of MSW landfills apparently contain information about the water transport through landfills analogous to the information provided by hydrographs from rivers on the characteristics of their catchment's basin (Holtan & Overton, 1963; Ogunkoya & Jenkins, 1993). Delayed time, shape and amplitude of leachate discharge peaks induced by heavy rainfall events allow at least first qualitative statements about water flow and water storage in landfills.

Leachate hydrographs of different landfills (Franzius, 1977; Ehrig, 1978; Jourdan, 1981; Åkesson & Nilsson, 1997; Döberl et al., 2002; Garcia de Cortázar et al., 2002) all show a quick response after heavy precipitation (that exceeds the retention capacity of the cover layers). Furthermore, a low discharge during dry periods without water input (evapotranspiration exceeds precipitation) is noticeable (see Figure 4-7, Figure 4-8 and Figure 4-9). This certain characteristic of leachate hydrographs is explained by opposed hydraulic properties of solid waste landfills. On the one hand water is retained and slowly released by capillary forces acting in micro-pores within the waste. On the other hand rapidly downward water flow occurs in connected macro-pores and fissures, which are caused by the coarse grading of MSW or differences in landfill settlement. The phenomenon of preferential flow is additionally concentrated by construction elements with high permeability such as gas wells. Furthermore, zones with less compaction or boundary zones represent favored areas for rapid water flow.

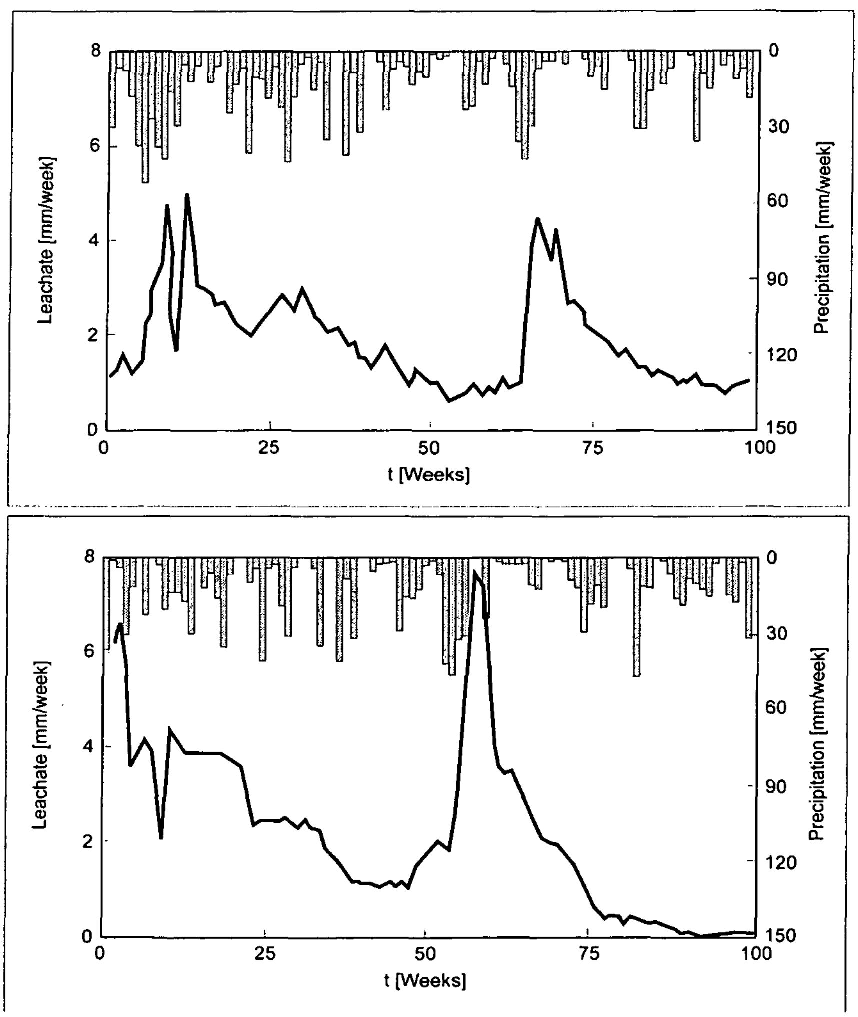



*Figure 4-7* ***Leachate hydrographs of German landfills (Ehrig, 1978)***

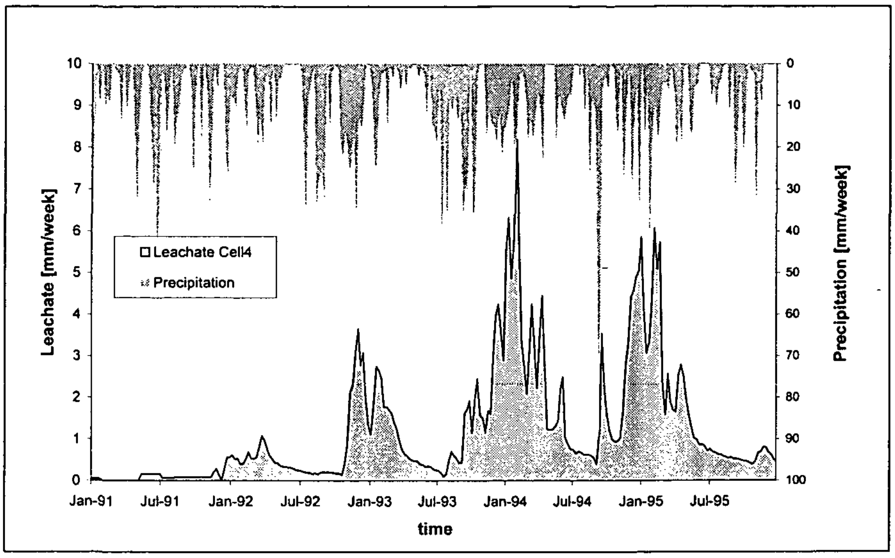



*Figure 4-8* ***Leachate hydrograph - Spillepeng test Cell 4 (Åkesson & Nilsson, 1997)***

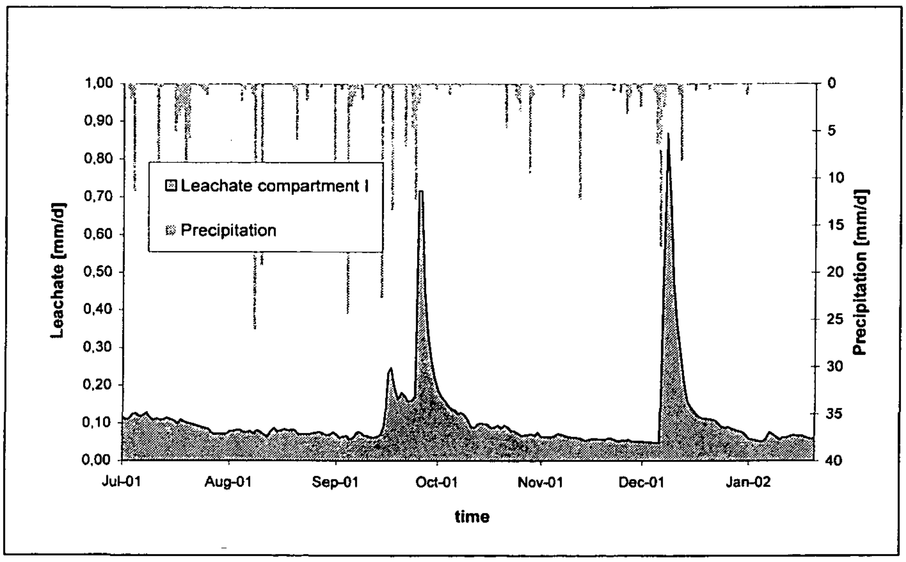



*Figure 4-9* ***Leachate hydrograph - Breitenau landfill Compartment I***

## *4.4. Tracer experiments*

In the field of hydrology tracer experiments are conducted to investigate the flow of water through certain systems (e.g. aquifers, soils, karst formations). Thereto the transport media water is charged with a tracer substance at a defined point (feeding point) and in another point (sampling point) the concentration of the tracer substance is recorded over the time. The direct outcome of a tracer test is the so called breakthrough curve (BTC), a chart containing tracer concentration versus time (Figure 4-10). The breakthrough curve enables to characterize the hydraulics of the investigated system.

At sanitary landfills only a few tracer experiments were reported in the literature (Maloszewski et al. 1995; Baumann & Schneider, 1998; Bendz & Singh, 1999; Beaven et al., 2001; Johnson et al., 1998; Rosqvist et al., 2001; Döberl et al., 2002).

Maloszewski et al. (1995) studied the water transport through waste lysimeters and found that up to 40% of heavy precipitation events drains off directly, i.e. within few weeks. Baumann & Schneider (1998), who investigated the water flow through a full sized landfill, came to a similar result. They noticed that 1/8 to 3/8 of the water input reaches the leachate collection system at the landfill bottom with negligible delay. Extensive analysis (Bendz & Singh, 1999; Rosqvist & Destouni, 2000; Rosqvist et al., 2001; Fourie et al., 2001) of tracer substances within waste bodies of different size (0.14 – 545 m$^3$) resulted in the conclusion that only a small fraction of water stored inside the waste takes part in the transport of solutes. Rosquist & Destouni (2000) calculated for an experimental landfill with an average height of 4 meter a mean residence time for the tracer substance of 20 days. The total amount of recovered tracer was around 34 %. The water content actively participating in the transport processes was quantified to be in a range of 6 to 12 %. Figure 4-10 gives an example of breakthrough curves for tracer tests that were performed at small waste columns. The shape is positively skewed with a long right hand tail, indicating a non-uniform transport of the solute through the waste mass. The early peak is attributed to rapid solute transport in favored flow paths (macro-pores) and the prolonged tails indicate slow water flow in less mobile domain (micro-pores).

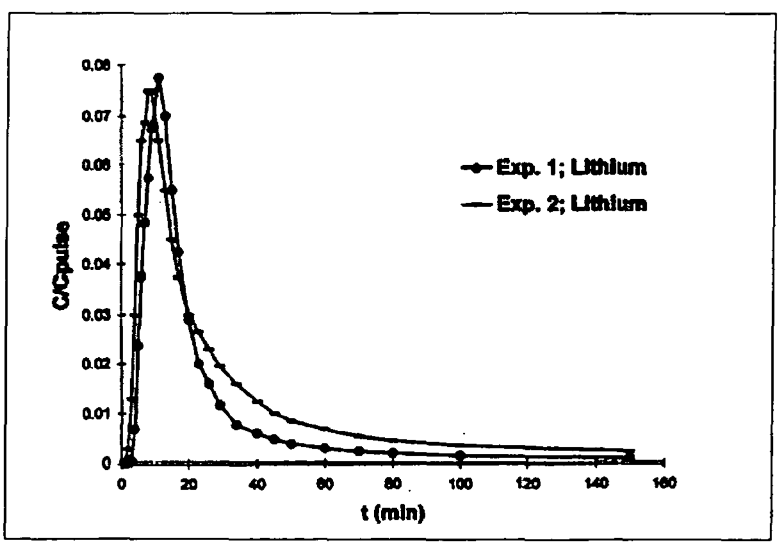



***Figure 4-10 Breakthrough curve of tracer tests (Rosqvist & Bendz, 1999)***

Beaven et al. (2001) performed a tracer test at a full-scale landfill, whereby the experiment was carried out under fully saturated conditions and the main flow direction of water was horizontally. This was due to the fact that no leachate collection system existed at the landfill bottom. The investigations indicate a drainable porosity taking part at the solute transport of around 3 %.

Results of Johnson et al. (1998) showed that even in more homogeneous landfills of bottom ash (compared to MSW) preferential flow paths play an important role for the hydrology.

Döberl et al. (2002) calculated for the Breitenau landfill (Compartment I with 12 m height) a mean tracer resistance time of 90 days and a water content participating in transport of 0,08 %. The recovery rate of the tracer substance however, was less than 20 % for this experiment (observation period: 260 days).

The phenomenon of rapid tracer breakthrough was noticed in all studies (e.g. Table 4-4) and attribute to preferential flow paths prevailing in landfills. The given breakthrough times expressed in bed volumes indicate the period till first significant appearance of tracer. One bed volume represents the total amount of water stored inside the landfill. Assuming completely uniform-flow conditions the breakthrough time of the tracer would be one bed volume.

***Table 4-4 Breakthrough times of tracer tests reported in the literature***

| Reference | Landfill volume [m³] | Landfill height [m] | Breakthrough time [bed volume] |
|---|---|---|---|
| Rosqvist et al. (2001) | 0.14 | 0.65 | ~ 0.68 – 0.87 |
| Bendz & Singh (1999) | ~ 3.5 | 1.2 | ~ 0.05 – 0.1 |
| Rosqvist & Destouni (2000) | ~ 545 | 4 | ~ 0.22 |
| Beaven et al. (2001) | ~ 15,000 | 20 | ~ 0.011# |
| Baumann et al. (1998) | ~ 50,000 | 6 | ~ 0.006 |
| Döberl et al. (2002) | ~ 30,000 | 12 | ~ 0.0008 |

*# Predominately horizontal flow under saturated conditions*

The data of Table 4-4 shows that the water flow becomes more non-uniform with rising volume and height of the landfill. That is demonstrated by declining breakthrough times. The assertion that the water transport in landfills is more non-uniform with rising depth was confirmed by Rosqvist et al. (1997), who noticed acceleration in the solute velocity towards the bottom of an experimental landfill. Taking the reduced hydraulic conductivity in greater depths into account (e.g. Bleiker et al., 1995; Powrie & Beaven, 1999) an increase in the solute velocity must attribute to heightened preferential flow. This finding is contrary to assertions of different studies (e.g. Bendz, 1998; Zeiss, 1997) that assume analogous to soil a more uniform water flow in greater depths.

## *4.5. Water flow pattern and its implication for water flow modeling*

The importance of channel flow for the hydrology of landfills have been identified by several researches (e.g. Ham & Bookter, 1982; Ehrig; 1983; Zeiss & Major, 1993) and already partly implemented into concepts for modeling water routing in landfills (Young & Davis, 1992; Uguccioni & Zeiss, 1997; Bendz, 1998; Obermann, 1999). As described in chapter 3.1.1.4 the waste mass in these models is split into a channel domain with rapid water flow surrounded by a matrix domain with slow water movement. Although the mathematical approaches describing the water transport in the two domains are diverse, all developed concepts are derived from the framework that has been carried out for non-uniform water flow in soils. Compared to soils, landfills are more heterogeneous, which results in a bigger fraction of preferential flow. This fact was taken into account by adjusting decisive parameters. The underlying assumption, the similarity between water flow in landfills and soils has not been justified or even discussed yet. The following section will point out the flow pattern in landfills on a macroscopic scale and will compare it with the non-uniform water flow in cracked or fissured soils.

As mentioned above landfilled MSW (due to its origin and its composition) is a highly heterogeneous media. The hydraulic behavior of single waste components ranges from highly water adsorbent to water repellent, from impermeable to well permeable materials. Nevertheless, it is possible to determine hydraulic parameters (chapter 4.2) for landfilled waste, if the investigated volume is big enough to be representative for the mixture of materials.

Considering the whole landfill body in contrast to the waste material itself, the degree of heterogeneity is increasing and therefore also representative volumes (Bear, 1972). This fact implicates that parameters determined at a small scale are invalid to characterize the whole landfill mass. In particular construction elements (e.g. gas wells or daily cover layers), areas with low mechanical compaction and boundary zones are responsible for the enhanced heterogeneity. Furthermore, landfilling and compaction of waste in thin layers leads to a horizontal stratification within the landfill. Consequently, the hydraulic conductivity shows a distinctly anisotropic behavior. Powrie & Beaven (1999) observed 10 times higher conductivity in the horizontal direction. Thus, a major part of the water flow through landfills occurs horizontally (Burrows et al., 1997). Additionally, the anisotropic behavior is increased due to the horizontal orientation of impermeable materials such as plastic sheets. Water is retained above these impervious surfaces inside the landfill body. Hanging water tables in different depths inside landfills, which have been reported in several investigations (e.g. Stegmann, 1990; Riehl-Herwirsch et al., 1995), attribute to those barriers caused by compaction and impermeable sheets, respectively. The retained water is forced to continue its flow in a more or less horizontal direction. Vertical channels and fissures resulting from the heterogeneous nature of the waste itself, from differences in landfill settlements, and from vertical construction elements with high permeability, short the horizontal pathways, and enable fast downward water transport inside landfills. The impervious surfaces (e.g. straightened plastic sheets) serve as water suppliers for the preferential flow paths. The described mechanism, water retaining and horizontal flow towards vertical channels, is repeated within every waste layer. This leads inevitably to a funneling of water in preferential flow paths with increasing depth. Subsequently, water flow becomes more non-uniform towards the landfill bottom. This predication is in agreement with investigations carried out by Wiemer (1982), Gabr & Valero (1995), Öman et al., (1999) and Yuen et al., (2001), who noted higher spatial differences in the water content towards the landfill bottom. Ziehmann et al. (2003), who studied the spatial difference in water supply of leachate collection systems, confirmed the existence of a highly non-uniform water flow at the landfill bottom.

Figure 4-11 provides a schematic for the water flow pattern inside MSW landfills. The picture is adopted from Mesu (1982), who first pointed out the importance of impermeable layers for the water movement. He compared the water transport inside landfills with the water flow from roofs.

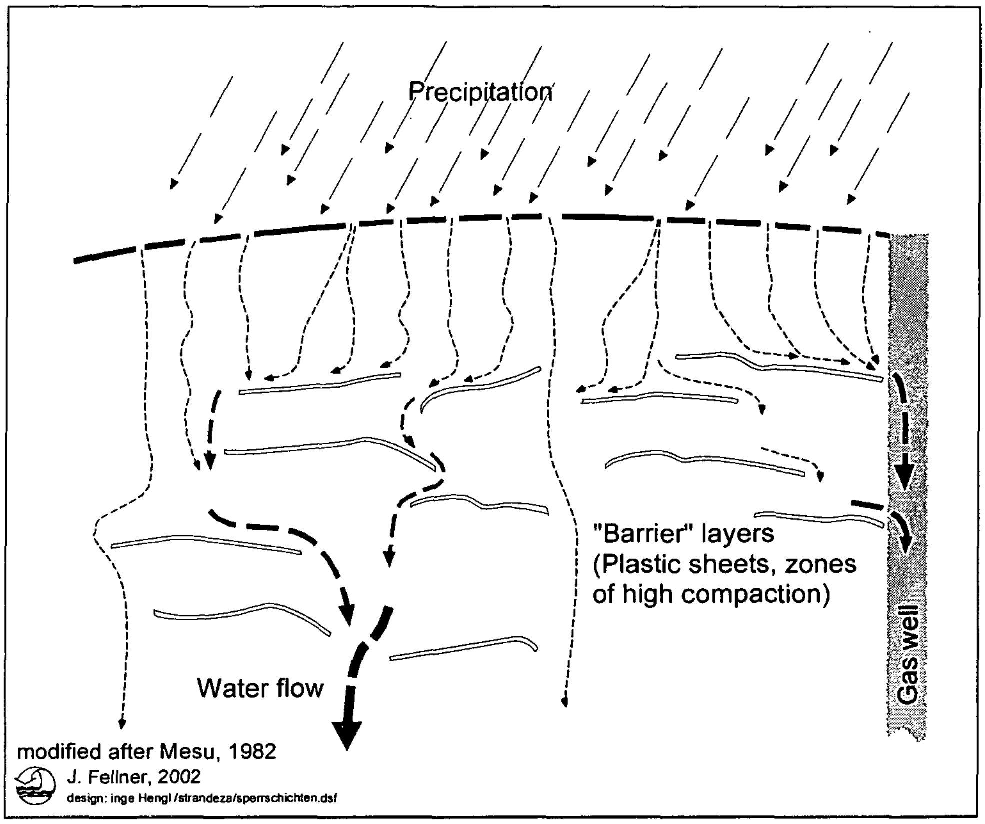



*Figure 4-11*      ***Water flow pattern in MSW landfills (modified after Mesu, 1982)***

The general water flow pattern in landfills is mainly determined by the structure of the landfill (e.g. impermeable horizontal surfaces, vertical channels). The portion of channel flow however and the matrix flow is not only dependent on the structure itself but also on the water application rate. New flow channels may develop or can be reached due to higher backwater above the impermeable surfaces during periods with high infiltration rates (Jasper et al., 1985). Even during dry periods (no additional water input) water is retained above plastic "barriers" and supplies the preferential flow paths.

Heterogeneous (fissured) soils, which are usually used as a comparable media for MSW landfills, show a different hydraulic behavior (Figure 4-12). No vertical flow limitation comparable to impermeable sheets is found in soils. Thus, water flow mainly occurs in the vertical direction. The application rate of water plays an important role for the water movement and its distribution in soils (Germann & DiPietro, 1996). During dry spells water transport is limited to the soil matrix only, whereas under wet conditions (during or short time after water infiltration events) macro-pores and fissures also contribute to the downward water movement. The soil matrix sorbs some of the water bypassing in preferential pathways, since capillary forces are acting in micro-pores. The fraction of water infiltrated from the fissures into the matrix depends on the water content of the soil matrix.

The mechanism and the degree of preferential flow in landfills and soils are strongly diverse. Thus, the distribution and pattern of preferential flow is different. Investigations in soils (e.g. Bundt et al., 2000) show that the effect of favored flow paths is becoming minor with depth, while the non-uniformity of the water flow in landfills increases towards the bottom.

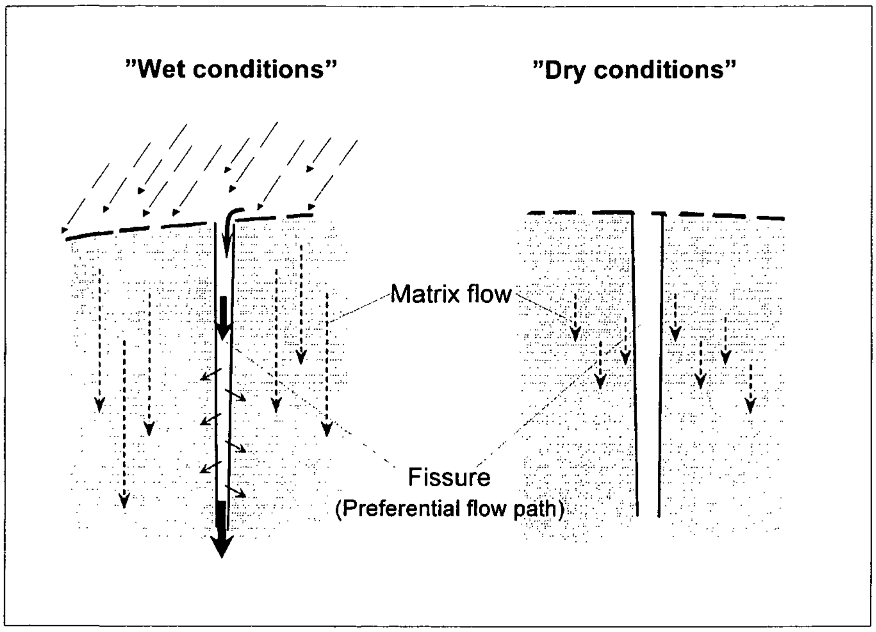



*Figure 4-12 Water flow in cracked soils during wetting periods and dry spells*

The main differences in the water flow between landfills and soils can be summarized as follows:

- in landfills the flow pattern mainly depends on the structure of the waste, whereas in soils the application rate of water is a decisive parameter determining the water flow
- contrary to soils, in landfills a large amount of water flow occurs in horizontal direction due to the anisotropic characteristic of landfills
- preferential flow occurs in soils only during wetting periods, while landfills show significant channel flow also during dry periods
- in soils the direction of water flow between the two domains is more or less restricted to flow from the channels into the fine pored matrix, while both flow directions are possible in landfills

- in soils the heterogeneity of the water flow decreases with depth, contrary to landfills where an increase of preferential flow in greater depths is observable
- finally the degree of non-uniformity of water flow in landfills is bigger than in soils

Due to the differences in the hydrology of landfills and soils, it can be concluded that present model concepts are based on inadequate assumptions. In particular the distinctive horizontal water flow in landfills (see Figure 4-11) caused by impermeable layers makes any abstraction into a one-dimensional vertical flow model insufficient. Even if a two-domain concept is realized, a major characteristic is neglected.

Based on conceptual considerations a simplification of the illustrated flow pattern (Figure 4-11) was proposed (see Figure 4-13). Analogous to previous modeling concepts (Uggucioni & Zeiss, 1997; Bendz, 1998; Obermann, 1999; Hartmann, 2000) the flow field is divided into a matrix domain with slow water transport and a channel domain with rapid water flow. Contrary to previous concepts however, the two-domain flow field is implemented in two dimensions. The matrix zone is characterized by low permeability and high water retention capacity, while the vertical channel domain shows high hydraulic conductivity and low (or even no) retention capacity. Thus, the matrix acts as storage zone for water and enables water release during dry spells. The channel domain allows fast downward water flow through the landfill and is responsible for the quick response of leachate discharge after precipitation.

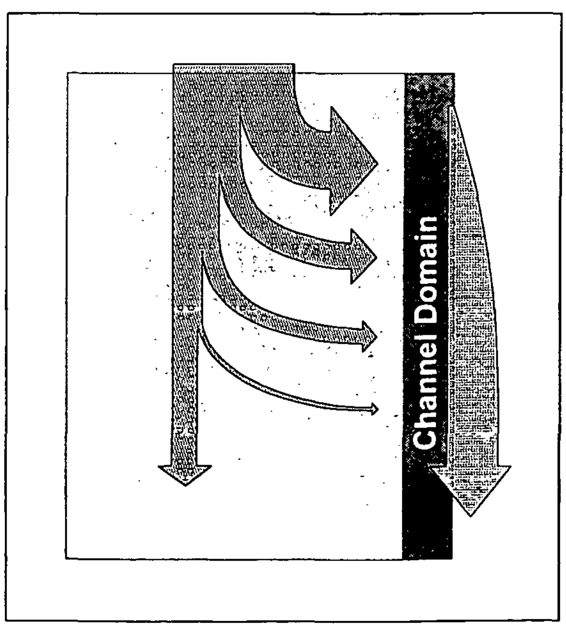



***Figure 4-13 Simplified water flow pattern in MSW landfills***

The spatial extent of the matrix zone is predominant. However, the channel domain is effective for the water transport. Near the surface water mainly flows in the matrix domain and the water distribution is more or less uniform. With increasing depth water is drained from the matrix domain into the channel, whereby the amount of water transferred from the matrix to the channel decreases with depth. At the landfill bottom a main fraction of water is originated from the preferential flow path. In consequence, the bulk of waste, which is by-passed by channel flow, increases towards the bottom.

### *4.6. Implementation of the new modeling concept into an existing software tool*

In principle the implementation of the flow pattern (displayed in Figure 4-13) into a mathematical transport model could be accomplished by two different options (Figure 4-14).

- The first manifest option (Figure 4-14 left side) to reproduce the proposed flow pattern is to introduce horizontal layers with low permeability. These layers retain water and enable the transport of water into the channel domain. However, quantitative information about their extent, permeability and inclination is lacking. Therefore modeling attempts would require several assumptions, and thus, extensive parameter calibration due to numerous unknown factors (hydraulic characteristics of matrix and channel domain as well as for each "barrier", spatial distribution, extent and inclination of "barrier" layers, and anisotropy of the hydraulic conductivity). A practical application of this option seems to be unworkable.
- The other possibility to ensure water flow from the fine-pored matrix domain into the preferential flow path according to the proposed flow scheme can be accomplished by defining the channel domain as a suction pipe, whereby the suction head is increasing towards the top of the landfill (Figure 4-14 right side). Higher suction head causes heightened water movement into the channel domain. The proposed concept reduces unknown characteristics of the impermeable layers to the suction head in the preferential flow path and the anisotropy of the hydraulic conductivity of the matrix domain.

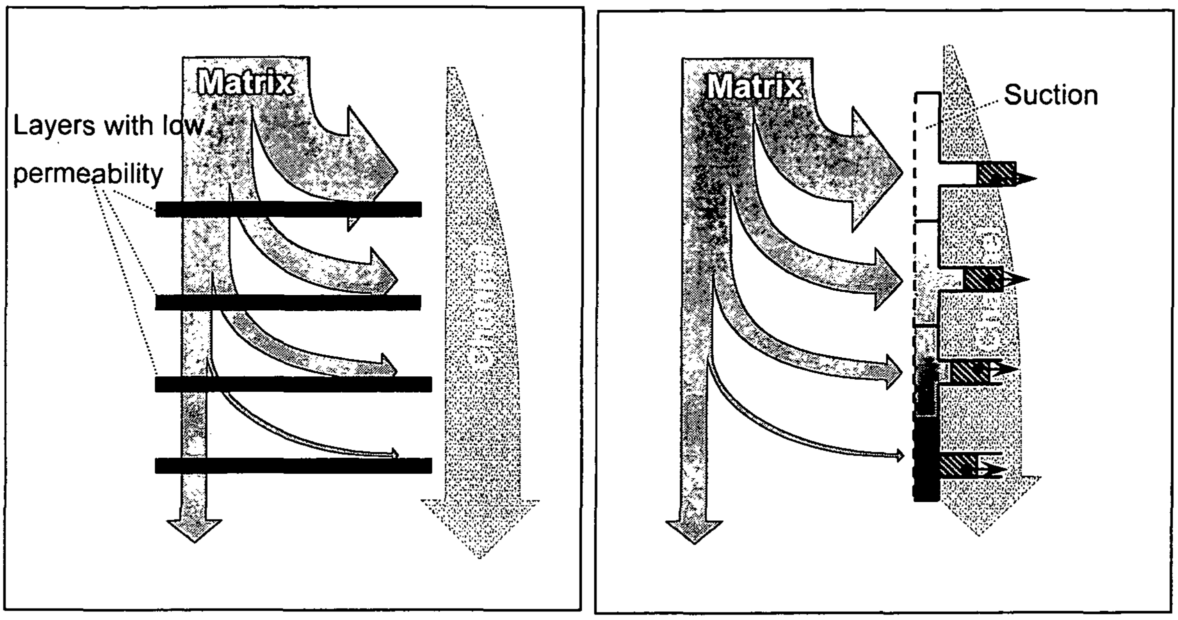



*Figure 4-14 Modeling options to implement the assumed flow pattern*

Due to the less unknown parameters and simpler practicability, the two-dimensional two-domain concept assuming suction power inside the channel is preferred. The uniformity respectively non-uniformity of the water flow can be controlled by the "properties" of matrix and channel domain.

The implementation of this option into an existing mathematical model for simulating water transport requires a tool including the following processes and abilities:

- two-dimensional simulation of water flow in variably saturated porous media
- spatial differences and anisotropy of the hydraulic characteristics within the two-dimensional flow field
- controllable suction power acting inside the preferential flow path

In addition appropriate software for simulating water flow through landfills must incorporate:

- the boundary conditions: seepage (at the bottom) and variable flux (at the top)
- water losses due to evaporation, evapotranspiration and surface runoff

Although numerous simulation tools (e.g. MODFLOW, Hill et al., 2000; HST3D, Kipp, 1997) of varying degree of complexity have been developed so far to quantify water flow in porous media, most programs were designed for groundwater flow, and thus, consider saturated conditions only. Two dimensional water transport under variably saturated

conditions is considered in e.g. SUTRA (Voss, 1984), VS2DI (Healy & Ronan, 1996), SEEP2D (GMS, 1996), FEMWATER (Lin et al., 1997), HYDRUS-2D (Šimunek et al., 1996). A comparison of these models shows that the above mentioned requirements for the two-dimensional two-domain concept are best fulfilled by the program HYDRUS-2D. It is the only model that enables a varying suction power (acting inside the channel domain) by applying the method of linear scaling of hydraulic properties (Vogel et al., 1991). HYDRUS-2D was developed at the U.S. Salinity Laboratory in Riverside to simulate water flow, solute and heat transport in variably saturated porous media at various boundary conditions. A short description of the software and its field of application are given in appendix 8.1.
The water flow in HYDRUS-2D is calculated using Richards equation (1931). This formula implies that gravity and capillary forces govern water flow. An assumption that concurs with the situation in the matrix domain quite well, but it is not applicative for the flow conditions in the channel domain. In fact it is contrary that, on the one hand large pores enable a rapid water flow and on the other hand capillary forces should dominate the water movement in the channel domain. However, this physical "error" is even required in order to realize the assumed flow pattern (Figure 4-14 right side), because water can only drain from the matrix if the capillary potential in the channel domain is lower. The introduction of capillarity acting in the preferential flow path represents from a physical point of view a suction pipe for the matrix domain, which enables that water is draining from the matrix domain into the channel.

Figure 4-15 (left side) shows the graphical operators interface of HYDRUS-2D with a defined two-dimensional two-domain flow field. Ascertaining hydraulic properties of matrix and channel domain as discussed above (high retention capacity and low permeability for the matrix domain and low retention capacity and high permeability for the channel domain) results in the given pressure (suction) head distribution (Figure 4-15 right side), a snap shot out of a simulation over a longer period. The dark colored zones indicate low suction head, whereas bright colors refer to areas with high suction head. The resulting pressure (suction) head distribution revealed by schematically implemented streamlines confirms that the new model concept enables to reproduce the investigated flow pattern.

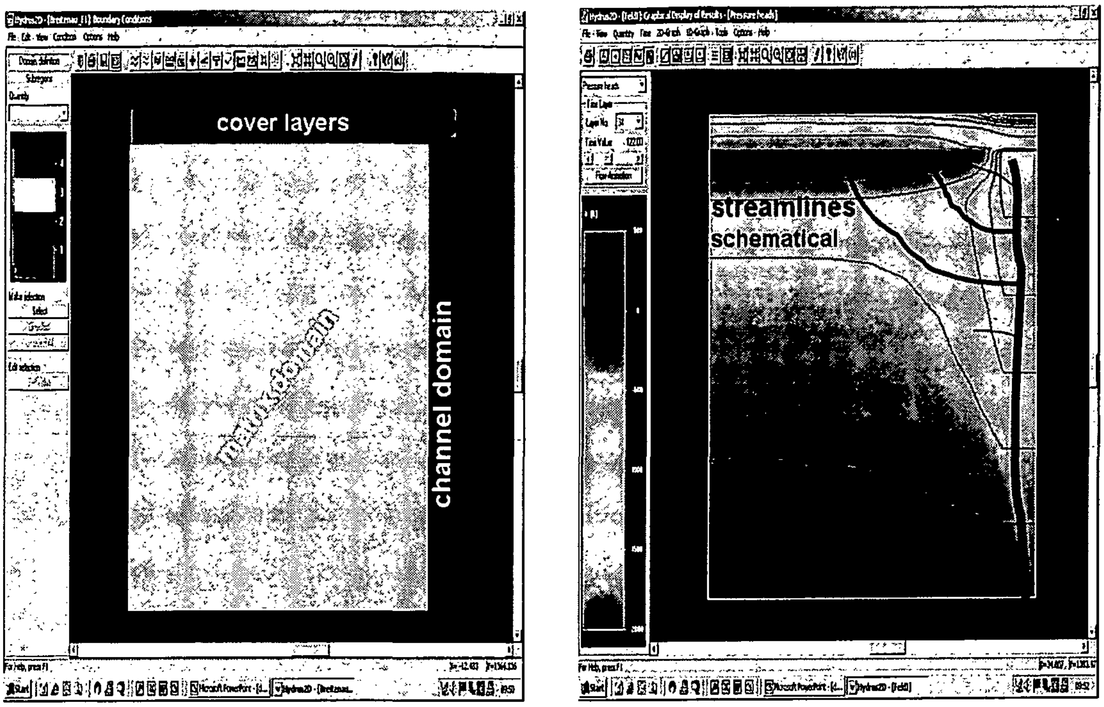



*Figure 4-15 Two-dimensional two-domain concept (left side) and resulting flow field (right side)*

## *4.7. Required input information of the new model*

The introduced two-dimensional two-domain water flow model based on HYDRUS-2D requires the following input information:

- "meteorological data":
  - precipitation
  - potential evapotranspiration
  - surface runoff
  - information about the vegetation cover
- flow field definition:
  - partitioning of matrix domain and channel domain
  - suction head (channel domain)
- hydraulic characteristics of the landfilled waste (matrix and channel domain) and the cover layers:
  - saturated/unsaturated hydraulic conductivity, anisotropy
  - water retention characteristics (van Genuchten model)
- initial conditions:
  - water content of the waste and the cover layers

### 4.7.1. Meteorological data

The meteorological data at the landfill site are required on a daily basis, whereby for HYDRUS-2D the potential evapotranspiration must be split up into evaporation and transpiration. This can be accomplished using information about the crop cover. Allen et al. (1998) proposes a linear relationship between crop cover fraction and the portioning into evaporation and transpiration (Figure 4-16).

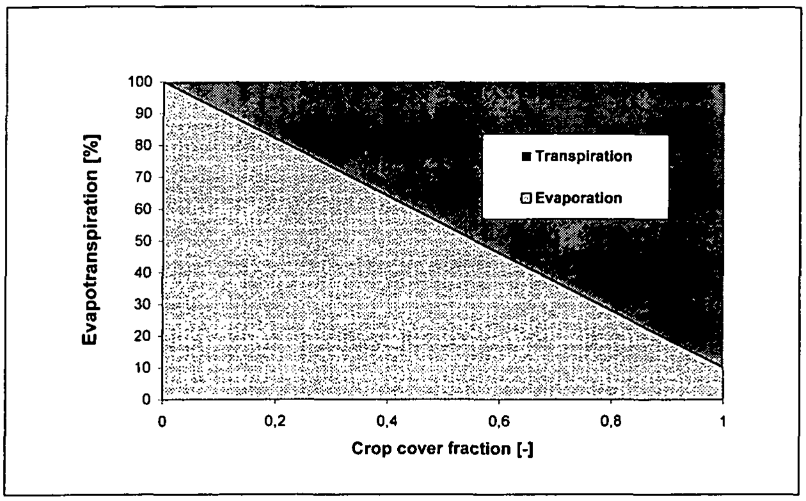



*Figure 4-16 Subdivision of evapotranspiration in dependence of the crop cover fraction*

In addition to the crop cover fraction, information about the root depth and the root distribution is needed to reproduce the hydrologic system landfill cover in a mathematical form.

### 4.7.2. Flow field definition

As described above, the proposed two-dimensional two-domain approach for modeling water flow in landfills postulates a separation of the waste mass into matrix and channel domain, whereby the area of the matrix zone is predominant. The proportion of the flow field is assumed as follows: 97 % matrix domain and 3 % channel domain. This partitioning is based on tracer experiments of Rosqvist (1999) and Beaven et al. (2001), who stated that the spatial fraction of preferential flow paths on the whole landfill body is less than 5 %. Additionally to the partitioning of the domains, the dimensions of the considered profile are of importance. For the simulations landfill profiles with less than 1 m width were assumed. The limiting to

1 m is based on two factors: On the one hand the limitation is necessary to curtail calculation times and on the other hand the limitation is based on a physical cause. Waste lenses inside the landfill reach a maximum horizontal length of 1 m (Bendz, 1998), which implies that preferential pathways can occur in this distance. In order to ensure alike proportions of the flow field, the width of the simulated landfill profile is altered with the landfill height. For a landfill of 10 to 12 m height a profile width of 1 m was chosen. This width reduces to 0.5 m for a landfill of 5 to 6 m height only.
One parameter introduced to accomplish and affect the presumed flow pattern is the scaling factor $\alpha_h$ of the pressure (suction) head (Vogel et al., 1991) in the channel domain. It allows confining the water flow pattern within a particular range.

### 4.7.3. Hydraulic characteristics of waste and cover layers

The hydraulic characteristics of cover layers can either be determined taking samples and performing laboratory test or using previously reported parameter values for similar soils.
In contrast, hydraulic parameters of MSW in particular of the two-domains (matrix and channel) are difficult to obtain. This is mainly due to two reasons. First, the two-domains are conceptual materials for the model approach rather than real existing media. The matrix domain however, can be understood as undisturbed waste mass. The second reason is that large representative volumes are necessary to characterize the mixture of materials. Therefore, hydraulic tests must be conducted at large samples. Furthermore, in previous studies reported parameters of MSW are hardly adjuvant. The values vary considerably (see chapter 4.2). Thus, reported hydraulic parameters provide only a first estimate for the parameter values of the matrix domain. The actual determination of the parameters must be performed during the calibration of the flow model.
For the channel domain a qualitative estimation of the hydraulic characteristics results in high hydraulic conductivity $K_s$ and low porosity n respectively saturated water content $\theta_s$.
Table 4-5 gives feasible ranges of different hydraulic parameters. A robust physical background of these values is missing, as these parameters are representing not only the media characteristics themselves but additionally interactions of the hydraulic system.

*Table 4-5* ***Feasible ranges of the hydraulic parameters of the two-domains using the van Genuchten model (1980)***

| "Material" | | Residual water cont. $\theta_r$ [-] | Saturated water cont. $\theta_s$ [-] | Form coefficient $\alpha$ [1/m] | Form coefficient $n_g$ [-] | Saturated conductivity $K_s$ [m/s] | Pore-connectivity l [-] |
|---|---|---|---|---|---|---|---|
| Matrix domain | min | 0 | 0.35 | 0.5 | 1.1 | $5\times10^{-8}$ | 2.0 |
| | max | 0.2 | 0.55 | 5.0 | 1.6 | $5\times10^{-6}$ | 40 |
| Channel domain | min | 0 | 0.005 | 0.5 | 1.2 | $1\times10^{-4}$ | 0.1 |
| | max | 0.001 | 0.05 | 5.0 | 3 | $1\times10^{-2}$ | 5.0 |

### 4.7.4. Initial conditions

Initial conditions required for the simulations with HYDRUS-2D are limited to the initial water content of the different "materials" (cover layers, matrix, and channel domain). The exact knowledge of the initial water content of the cover layers is less important, as no significant water storage over a longer time period (years) is possible within theses layers. In contrast the situation for the landfill body, in particular for the matrix domain storage processes over long periods are highly relevant. Therefore, a good estimation of the initial water content is crucial. Initial water contents vary less compared to other hydraulic properties of MSW. Table 4-6 gives an overview of published values. The majority of the reported mass water content values lie between 25 and 35 % (referred to wet mass of MSW).

*Table 4-6* ***Initial mass water content of MSW***

| Reference | Initial mass water content $m_w$ [m%] WS* |
|---|---|
| EAWAG (1975, cited in Brunner, 1976) | 30 |
| Spillmann & Collins (1986) | 26 |
| Baccini et al. (1987) | 30 |
| Ehrig (1989) | 30 |
| Reimann & Hämmerli (1995) | 12 – 35 |
| Schachermayer et al. (1994) | 27 – 30 |
| Fehringer et al. (1997) | 30 |
| Morf et al. (2003) | ~22 |
| Skutan & Brunner (2003) | 37.5 |

* *WS ... referred to wet mass*

In order to obtain the required input data for HYDRUS-2D the values must be converted from mass's percentage $m_w$ to volumetric water content $\theta$. Therefore, information on the density $\rho$ of the landfilled waste is required.

$$\theta = \rho \cdot m_w \qquad \textbf{\textit{Equation 4-2}}$$

| | |
|---|---|
| $\theta$ | *Volumetric water content [$m^3\ m^{-3}$]* |
| $\rho$ | *Wet density of the landfilled waste [$Mg\ m^{-3}$]* |
| $m_w$ | *Water content, referred to wet mass [$m^3\ Mg^{-1}$]* |

As mentioned above the water flow model must be calibrated and validated using data on the leachate discharge. The temporal resolution of theses data must be at least on a weekly basis. If only monthly values are available the model calibration and validation would be limited to water balance considerations only, as internal water flow processes can not be determined at this time scale. Calibration and subsequent validation of the flow model must be carried out using different time series of leachate discharge. The chronology of the data sets used is irrelevant. In order to ensure reliable calibration and validation, applied leachate records must show alternations in discharge.

The following parameters are predominantly determining for the introduced two-dimensional two-domain flow model and need to be adjusted during the calibration procedure.

- hydraulic characteristics of matrix domain and preferential flow path
- anisotropy, ratio between horizontal and vertical conductivity of the matrix domain
- scaling factor for the pressure (suction) head of the preferential flow path

Additionally the hydraulic parameters of the cover layers must be trimmed within a plausible range.

# 5. Simulation Results (Model Calibration and Validation)

The proposed two-dimensional two-domain model for calculating water flow through MSW landfills was validated using data from two landfill sites. For the first application data from the experimental landfill Breitenau (Riehl-Herwirsch et al., 1995) situated in Lower Austria was available. Observed leachate generation rates from this site were used to calibrate and subsequently validate the introduced model concept. The second calibration and validation of the model was conducted during a research visit at Lund University using information from landfill test cells at the Spillepeng site in Malmö, Sweden (Nilsson et al., 1992).
Both landfills are characterized by a field scale size and a good scientific documentation of operation parameters over a longer period.

## *5.1. Case study experimental landfill Breitenau*

### 5.1.1. Site description

The experimental landfill Breitenau is located at a former gravel mining site 60 km south of Vienna. Since it is one of the best documented landfills (Riehl-Herwirsch et al., 1995; Binner et al., 1997; Döberl et al., 2002), it represents a unique opportunity to investigate the behavior of organic "reactor" landfills. The site was filled up with around 95,000 tons of MSW in the years 1987 and 1988. The landfill is divided into three separate compartments of different size, with different capping systems and different base liner systems (see Figure 5-1). Compartment I (C I) is covered with a thin gravel layer (0.2 m) and above a silt layer of around 0.9 m. Approximately 35,000 tons of MSW have been landfilled in this compartment. Compartment II (C II) contains 25,600 tons of waste. The cover of this field is not uniform. Half of C II is only covered by gravel (1.3 m), whereas the other half has an additional layer of compost (0.7 m). At Compartment III (C III) the same capping system (gravel and compost) as at the second half of Compartment II was installed. The landfilled waste at C III amounts 33,200 tons. Mineral dams and geomembranes separate the three compartments. The base liner system of C I and C II consists of a mineral liner (1.4 m silt) in combination with a geomembrane for control measurements. At Compartment III a sealing made up by a geomembrane was installed.

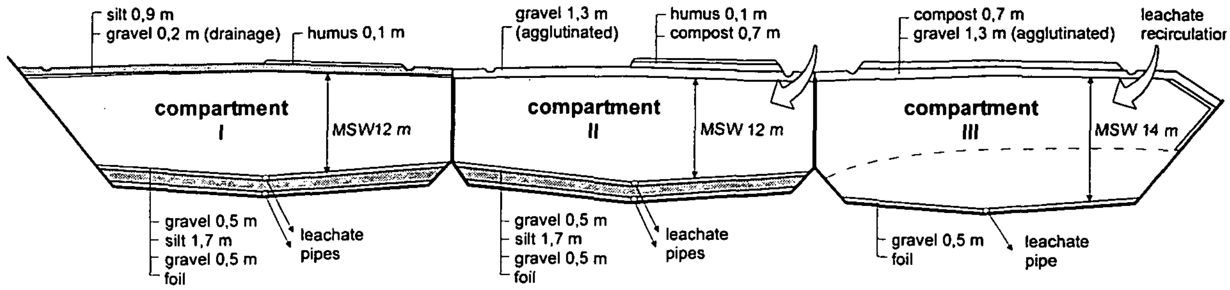



*Figure 5-1 Cross-section of the landfill Breitenau (Huber et al., 2004)*

Drainage layers of coarse gravel were placed above the base lining systems. These layers are connected to drain pipes which lead to the leachate collection house at the landfill base. In the collection house, the discharged water is collected for each compartment separately in tanks and then pumped to the public sewage system. Since the operation of the landfill in 1987, the leachate discharge has been measured using different methods resulting in data of different temporal resolution. In the first years after landfilling (till July 1997) differences in the water level of the leachate collection tanks were used to determine the outflow. From July 1997 to June of 2001 leachate generation was measured using mechanical gauges. Finally in July 2001 the discharge registration method was upgraded to electromagnetic flowmeters in combination with data loggers. These devices enable to record leachate outflow with high temporal resolution (e.g. registration interval of ten minutes).

The purpose of the initial project "Hausmüllversuchsanlage Breitenau" (Riehl-Herwirsch et al., 1995) was to evaluate the influence of waste filled gravel pits on the groundwater. In particular the suitability of sludge derived from gravel washing plants as a barrier between groundwater and waste was investigated. Additionally, the influence of different cover layers on the water balance was studied. It was shown that the capping system built up by compost and gravel (Compartment III) results in least leachate. Whereas the mineral top sealing system with silt (Compartment I) failed after two years, because an obvious increasing of leachate discharge after heavy precipitation events was noticed.

### 5.1.2. Input information

As stated in section 4.7 the introduced model requires information on the hydraulic characteristics of the cover layers and the waste domains (matrix and channel domain), and the meteorological conditions prevailing at the landfill site. In the following a brief summary of the required input data for the modeling effort at the landfill Breitenau is given.

#### 5.1.2.1. Meteorological data

- Precipitation:

The required input precipitation was derived using mean values of three meteorological stations (Neunkirchen, Saubersdorf and Wr. Neustadt) nearby the landfill site (Figure 5-3).

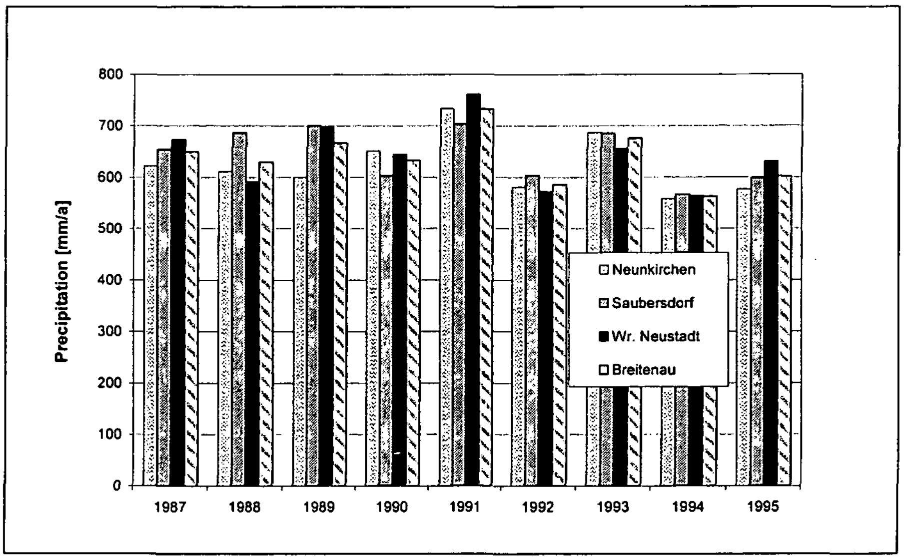



*Figure 5-2 Annual precipitation of the meteorological stations nearby the landfill Breitenau*

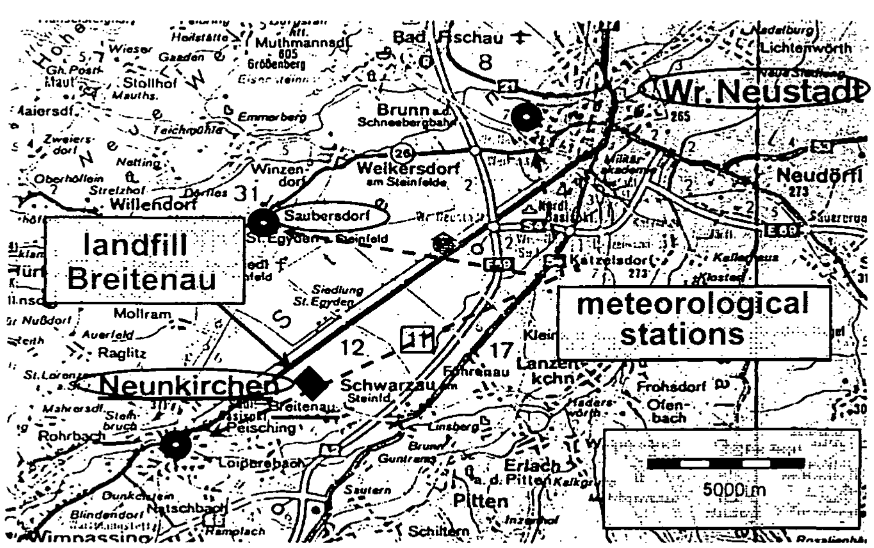



*Figure 5-3 Location of the landfill Breitenau and meteorological stations nearby*

- Evapotranspiration:

The rate of the reference evapotranspiration $ET_0$ at the landfill site was evaluated according to Haude (1954) using regionally adapted factors of phenology (Dobesch, 1991). The reference values represent the potential evapotranspiration $ET_p$ of grass of 12 cm height during the growing season. In order to obtain the potential evapotranspiration of a different vegetation cover, the reference values $ET_0$ are multiplied by a crop coefficient $K_c$. This coefficient is specific to the crop type and its developmental stage. For vegetation covers of numerous different crops an approximately linear relationship between crop coefficient $K_c$ and the crop cover fraction can be assumed (see Table 5-1 according to Allen et al., 1998).

***Table 5-1*** ***Relation between crop cover fraction and crop coefficient***

| Crop cover fraction [%] | Crop coefficient $K_c$ [-] |
|---|---|
| 100 | 0.95 – 1.15 |
| 75 | 0.75 – 0.85 |
| 50 | 0.55 – 0.65 |
| 25 | 0.4 – 0.5 |

Beyond the growing season and for bare soils, a dependency of the crop coefficient $K_c$ from the reference evapotranspiration $ET_0$ and the rainfall interval according to Allen et al. (1998) is assumed (Figure 5-4). The average interval between significant rainfall events was estimated to 7 days at the landfill Breitenau.

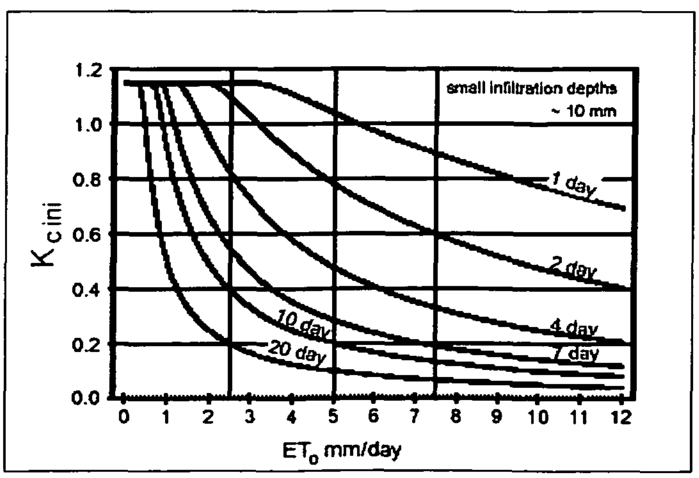



***Figure 5-4*** ***Relation between crop coefficient, reference evapotranspiration and rainfall interval (Allen et al., 1998)***

The required input parameters (air temperature and moisture content at 2 p.m.) for the calculation after Haude (1954) were derived from the meteorological station Wiener Neustadt, as it is the only station nearby measuring these parameters. Additionally to the computation after Haude (1954), evapotranspiration was estimated using the common method of Penman-Monteith (Bevan, 1979). Due to the lack of data on wind speed and sunshine hours (required for the calculation after Penman-Montheith) at the landfill site, feasible ranges for these parameters had to be assessed. Figure 5-5 gives a comparison of annual evapotranspiration values calculated according to Haude and Penman-Monteith assuming an average wind speed of 1 to 2 m/s (Riehl-Herwisch et al., 1995) and a relative sunshine duration of 0.42 (ZAMG, 2002). The evapotranspiration values determined with different methods match within the domain of uncertainty (caused by estimations regarding wind speed and sunshine hours).

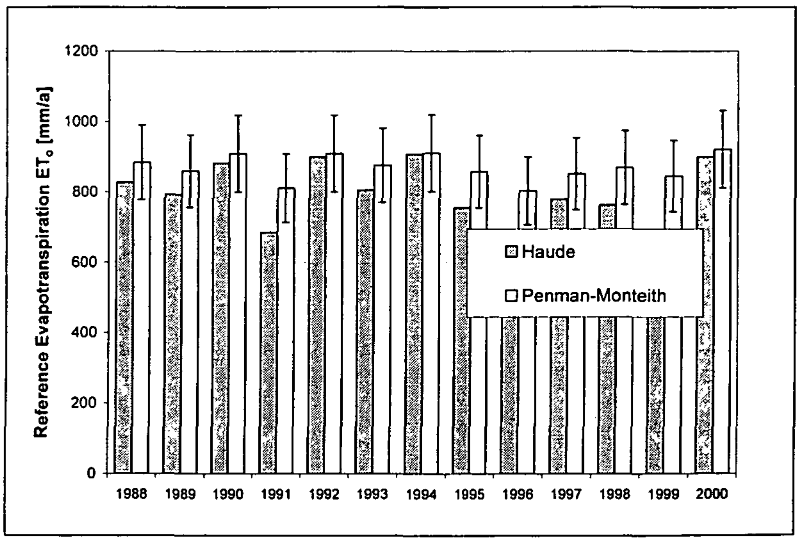



***Figure 5-5*** ***Comparison of annual reference evapotranspiration (after Haude and Penman-Monteith)***

The method after Haude was preferred to determine the reference evapotranspiration at the landfill site, since some of the parameters required for the Penman-Monteith equation are not available or only in form of annual averages. Small deviations between the results (Figure 5-5) of both methods show that this approach is adequate.

Information on the runoff from the landfill exists only till December 1991 in the form of monthly values. Settlements of the landfill surface led to changes in the general slope direction (lowest point of the surface in the center of the compartments), which made overland flow out of the compartments impossible.

As mentioned above, the generation of leachate was measured at different intervals. Since the calibration and validation of the model requires data with high temporal resolution, only observed discharge values since June 2001 are practical for this purpose.

#### 5.1.2.2. Hydraulic properties

Additionally to meteorological data the landfill model requires information on the hydraulic properties of the cover layers and the landfilled waste (matrix domain and channel domain). The cover materials of the Breitenau landfill were characterized by the Institute of Hydraulics and Rural Water Management at the University of Natural Resources and Applied Life Sciences, Vienna (Loiskandl, 2001). Saturated hydraulic conductivity, grain size distribution, bulk density and porosity of the different materials have been determined. A summary of the results is presented in Table 5-2 and Table 5-3.

***Table 5-2 Hydraulic conductivity, porosity and density of the landfill cover layers (Breitenau)***

| Cover material | Saturated hydraulic conductivity k [cm/d] | Porosity n [-] | Bulk density $\rho_d$ [Mg/m³] |
|---|---|---|---|
| „Silt“ | 90 – 250 | 0.25 – 0.37 | 1.66 – 2.05 |
| „Gavel“ *agglutinated* | 2 – 40 | 0.27 – 0.29 | 1.98 |
| “Gravel” *drainage layer* | 500 – 1,500 | – | – |
| „Compost“ | 200 – 550 | 0.66 – 0.77 | 0.53 – 0.78 |

***Table 5-3 Grain size distribution of the landfill cover layers (Breitenau)***

| Cover material | Coarse grit (> 2mm) | Fine grit (< 2mm) | Sand (<2mm) | Silt (<63µm) | Clay (<2µm) |
|---|---|---|---|---|---|
| | [Bulk-%] | | | | |
| „Silt“ | 56 – 64 | 36 – 44 | 40 – 42 | 40 – 43 | 15 – 19 |
| „Gravel“ *agglutinated* | 74 – 82 | 18 – 26 | 55 – 59 | 27 – 33 | 12 – 14 |
| „Compost“ | 51 – 66 | 34 – 49 | 62 – 71 | 23 – 30 | 4 – 8 |

By applying so called pedo-transfer-functions PTF, information about the grain size distribution was combined with the porosity and the bulk density to estimate the water retention characteristics of the different materials. In particular the software Rosetta (Schaap et al., 2001) was applied to assess probable parameter values for the retention model of van Genuchten (1980) which is used in HYDRUS-2D. This model consists of empirical equations that describe the relationship between water content and pressure head (equations see Appendix 8.2). The results of Rosetta, possible ranges of van Genuchten parameters for the different cover layers, are presented in Table 5-4.

***Table 5-4*** ***Ranges of van Genuchten parameters of the landfill cover materials (after Rosetta)***

| Cover material | Method | van Genuchten parameter | | | |
|---|---|---|---|---|---|
| | | Residual water content $\theta_r$ [-] | Saturated water content $\theta_s$ [-] | Parameter $\alpha$ [1/m] | Exponent $n_g$ [-] |
| „silt“ | PTF (Rosetta) | 0,03 – 0,05 | 0,23 – 0,34 | 1,5 – 4,2 | 1,16 – 1,38 |
| „Gravel“ *agglutinated* | PTF (Rosetta) | 0,03 – 0,04 | 0,22 – 0,27 | 4 – 7 | 1,10 – 1,66 |
| „Compost“ | PTF (Rosetta) | 0,04 – 0,07 | 0,50 – 0,65 | 1,5 – 4 | 1,26 – 1,46 |

The "definite" hydraulic properties of the waste domains (matrix and channel) need to be determined by calibrating the landfill model, whereby the parameters are varied within the ranges given in Table 4-5. Additionally, the parameters characterizing the cover layers (Table 5-4) are adjusted during calibration.

### 5.1.3. Results of Compartment I

The model was calibrated using leachate data from June 2001 till December 2001. This period was chosen in order to perform the calibration procedure at two peaks of leachate discharge that were induced by precipitation events. The different vegetation cover and thus different potential evapotranspiration within Compartment I was accounted for using an average value of evapotranspiration. The maximum depth of crop roots was set to 25 cm.

Figure 5-6 presents the simplifications of the hydraulic system for the modeling effort. To avoid unrealistic capillary water rise from the waste mass up into the landfill cover, simulations were conducted separately for the landfill cover and the waste body. Results (seepage) obtained from the system cover layers served as water input for the hydraulic system landfill that consists of matrix domain and preferential flow path.

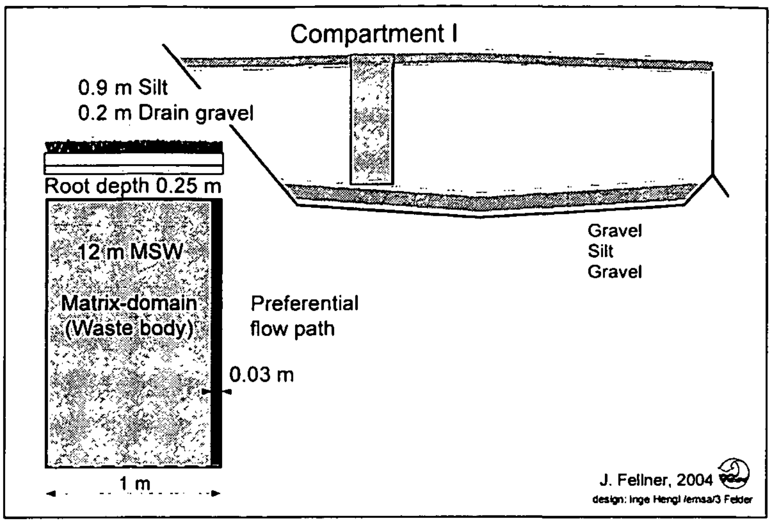



*Figure 5-6 Simplified model system (Compartment I)*

The model was calibrated using the method of trial and error. The match between simulated and observed leachate discharge was predominantly evaluated by graphical comparison. Finally a quantitative quality grade according to the Gaussian sum of square error (Hartung et al., 1993) was determined.

Applying the two-dimensional two-domain concept, it was possible to predict base flow during dry periods (unaffected by precipitation) as well as discharge peaks after heavy rainfall. The calibrated hydraulic parameters of the cover layers and the waste domains are presented in Table 5-5 and Table 5-6.

*Table 5-5 Water retention parameters for Compartment I (Breitenau)*

| Material | Water content | | Retention | |
|---|---|---|---|---|
| | Residual $\theta_r$ [-] | Saturated $\theta_s$ [-] | Coefficient $\alpha$ [1/cm] | Exponent $n_g$ [-] |
| Silt | 0.06 | 0.22 | 0.015 | 1.18 |
| Gravel (drain layer) | 0.04 | 0.15 | 0.145 | 1.5 |
| Matrix-domain | 0.15 | 0.5 | 0.02 | 1.4 |
| Channel domain | 0 | 0.01 | 0.02 | 1.4 |

*Table 5-6* ***Hydraulic conductivity parameters for Compartment I (Breitenau)***

| Material | Hydraulic conductivity | |
|---|---|---|
| | Saturated $K_s$ [cm/d] | Pore-connectivity l [-] |
| Silt | 200 | 0.5 |
| Gravel (drain layer) | 700 | 0.5 |
| Matrix-domain | 10 | 20 |
| Channel domain | 30,000 | 0.5 |

Additionally, the anisotropy of the matrix domain concerning the hydraulic conductivity $K^A_h$ (ratio between horizontal and vertical hydraulic conductivity $K_h/K_v$) had to be set to 2.0, and the scaling factor for the pressure head $\alpha_h$ of the preferential flow path to a value of 5. These values provided good agreement between simulated and observed leachate discharges (Figure 5-7).

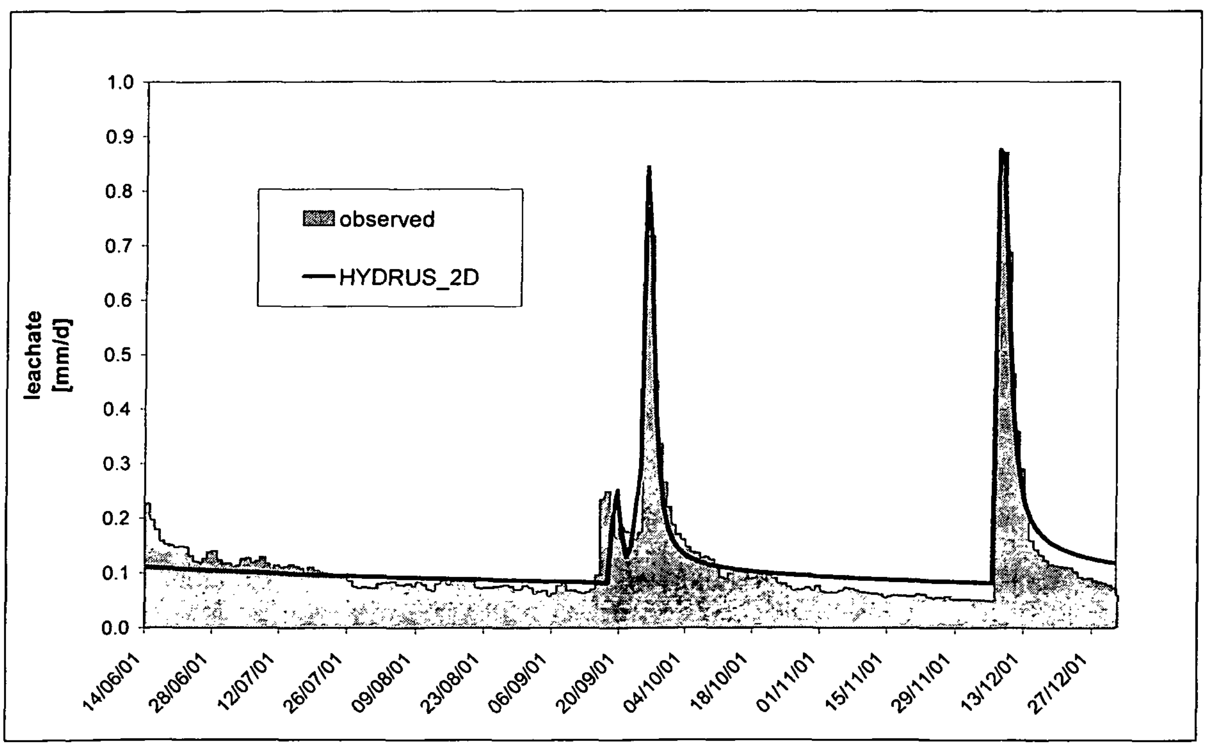



*Figure 5-7* ***Observed and simulated leachate discharge for the calibration period (Compartment I)***

The quality grade (according to Gaussian sum of square error) for the calibration period gives a mean discrepancy of 0.03 mm/d between observed and predicted data.

The extrapolation of the model shows that predicted and observed leachate discharges are close even beyond the calibration period (Figure 5-8). This is remarkable since maximum flow rates are nearly 10 times higher during the validation period compared to the discharge rates used for the calibration. The mean deviation of the prediction from the observed discharge is around 0.25 mm/d. Differences between model results and observation (February and May 2002) may attribute to uncertainties of the meteorological data, as measurements from nearby stations and not from the landfill site itself served as input data.

Figure 5-9 compares calculated and measured cumulative discharge during the period from June 2001 till July 2002. The model predicts a total leachate amount of 105 mm which is close to the observed value of 101 mm. The maximum error did not exceed 17 % of the observed discharge.

In general it can be postulated that the water flow model was validated successfully at Compartment I.

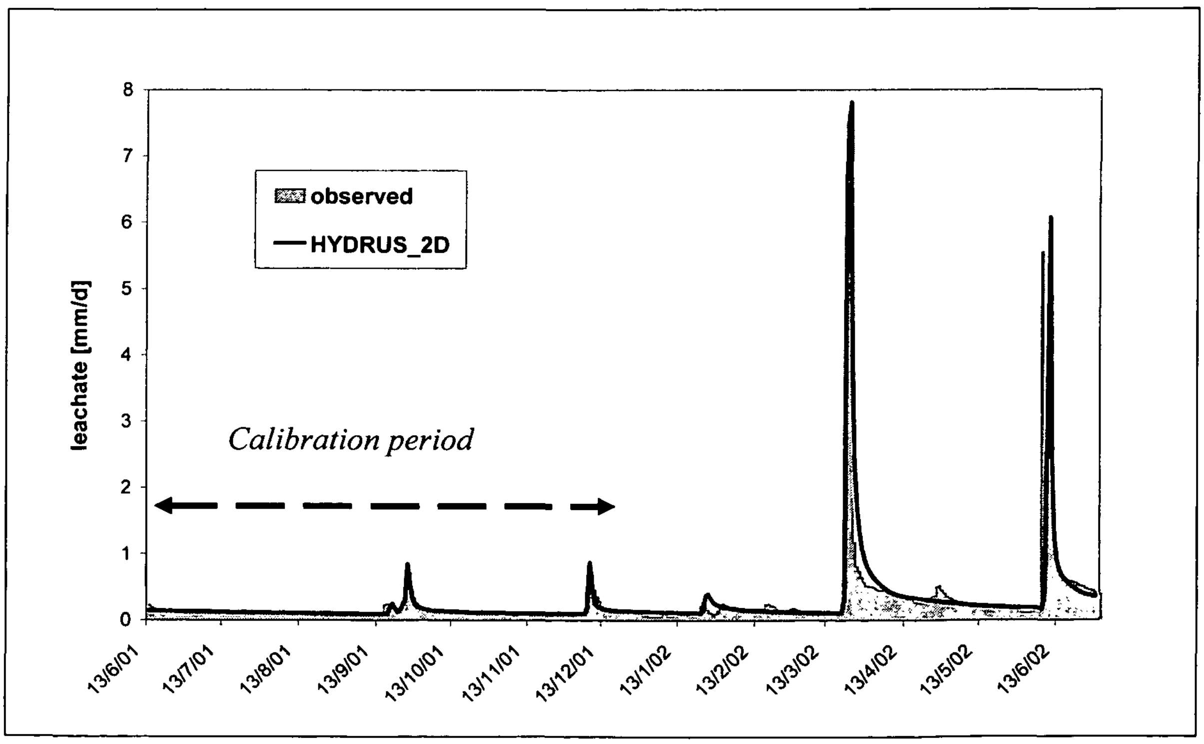



***Figure 5-8 Observed and simulated leachate discharge (Compartment I)***

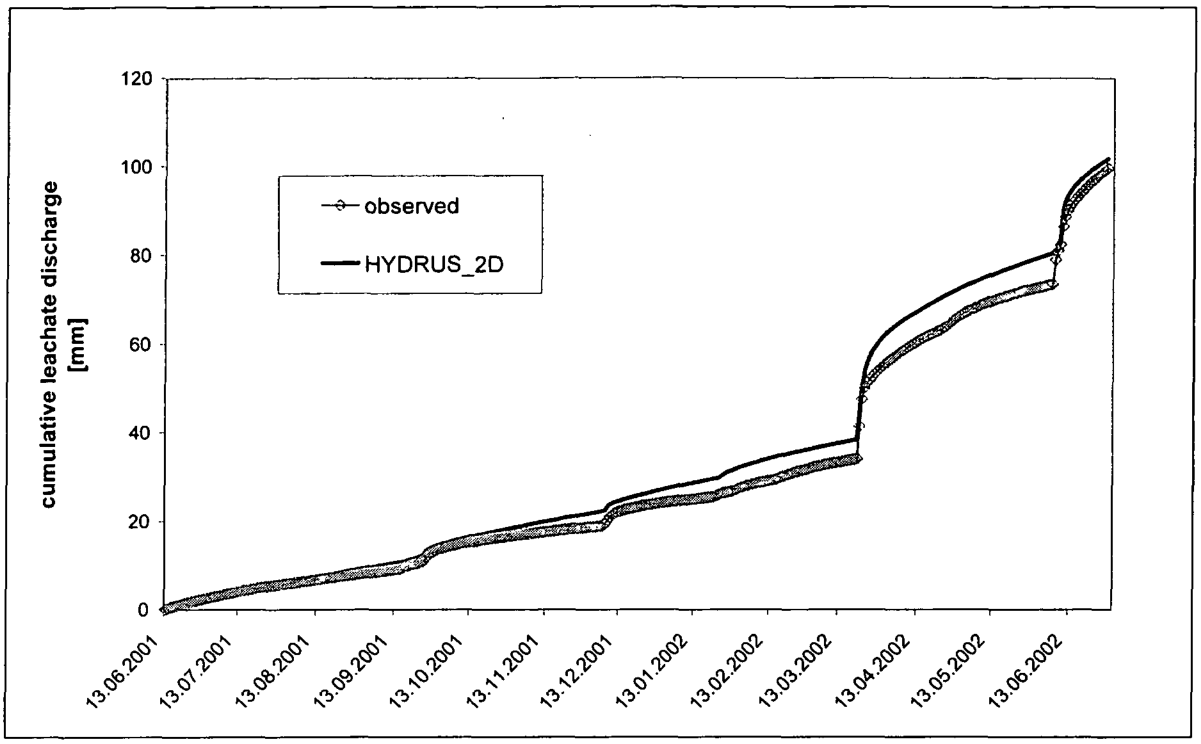



***Figure 5-9 Observed and simulated cumulative leachate discharge (Compartment I)***

### 5.1.4. Results of Compartment II

For Compartment II two separate water flow simulations had to be conducted due to the different capping systems within this compartment (see Figure 5-10). The results of the simulations were weighted according to the surface areas of the different cover layers and added up. This procedure made the calibration of the water flow model extremely difficult and time consuming, as more parameters had to be adjusted and probably several calibration optima exist. The calibration period had to be extended (compared to C I) till the end of June 2002, since no influence of precipitation on the leachate discharge was observable until spring 2002. Thus, the whole available time series was required for the calibration of the model. The lack of a further data set made it impossible to further validate the model.

The maximum root depth of the vegetation was assumed to be 15 cm and 45 cm at Compartment II/1 (gravel surface) and II/2 (compost surface), respectively.

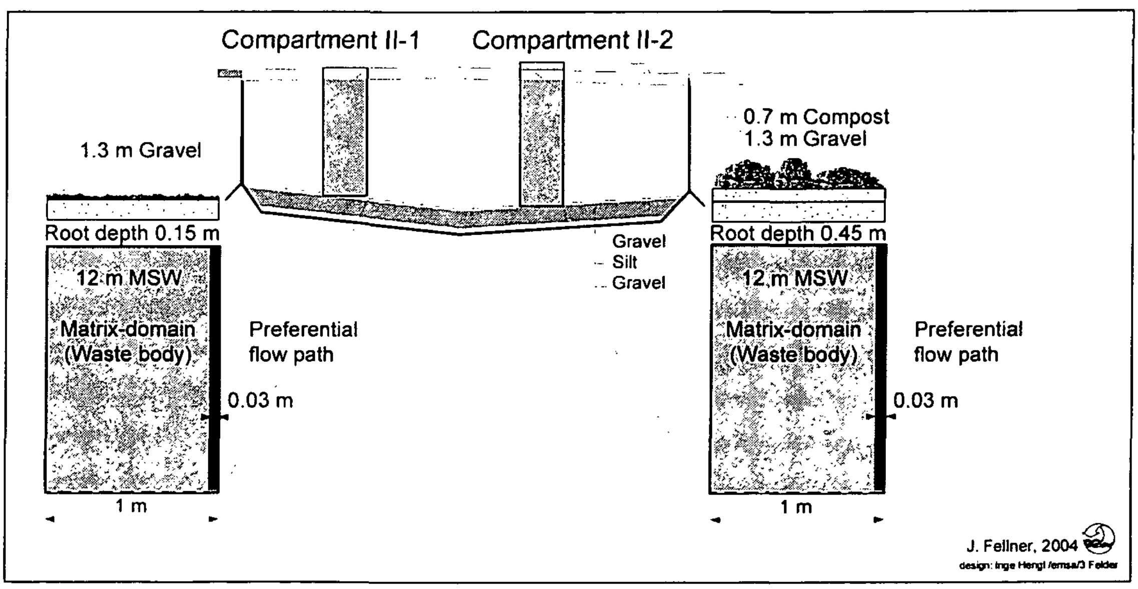



*Figure 5-10 Simplified model system (Compartment II)*

Within the scope of model calibration the following values for the hydraulic parameters of the different "materials" have been determined:

*Table 5-7 Water retention parameters for Compartment II (Breitenau)*

| Material | Water content | | Retention | |
|---|---|---|---|---|
| | Residual $\theta_r$ [-] | Saturated $\theta_s$ [-] | Coefficient $\alpha$ [1/cm] | Exponent $n_g$ [-] |
| Compost | 0.20 | 0.5 | 0.015 | 1.2 |
| Gravel (agglutinated) CII/1* | 0.03 | 0.29 | 0.03 | 1.5 |
| Gravel (agglutinated) CII/2* | 0.04 | 0.23 | 0.04 | 1.5 |
| Matrix domain | 0.15 | 0.5 | 0.02 | 1.4 |
| Channel domain | 0 | 0.01 | 0.02 | 1.4 |

*Table 5-8 Hydraulic conductivity parameters for Compartment II (Breitenau)*

| Material | Hydraulic conductivity | |
|---|---|---|
| | Saturated $K_s$ [cm/d] | Pore-connectivity l [-] |
| Compost | 300 | 0.5 |
| Gravel (agglutinated) CII/1* | 10 | 0.5 |
| Gravel (agglutinated) CII/2* | 7 | 0.5 |
| Matrix domain | 10 | 23 |
| Channel domain | 2,000 | 0.5 |

**Diverse parameter values for the gravel layer within Compartment II are explained by different degrees of agglutination. The gravel underlying compost at Compartment II/2 exhibits a higher level of agglutination and shows therefore less porosity and conductivity.*

Best match between observed and predicted leachate outflow was yielded setting the anisotropy of the hydraulic conductivity $K^A_h$ to 0.55 and the pressure head scaling factor of the channel domain to 1.9. Figure 5-11 presents measured and calculated leachate discharge versus time. The hydrographs coincide remarkably. The average deviation of the simulated water flow from the observed values was less than 16 % (according to Gaussian sum of square error). The difference refers mainly to a slight delay of the simulated discharge peaks, as the cumulative outflow values (Figure 5-11) agree well.

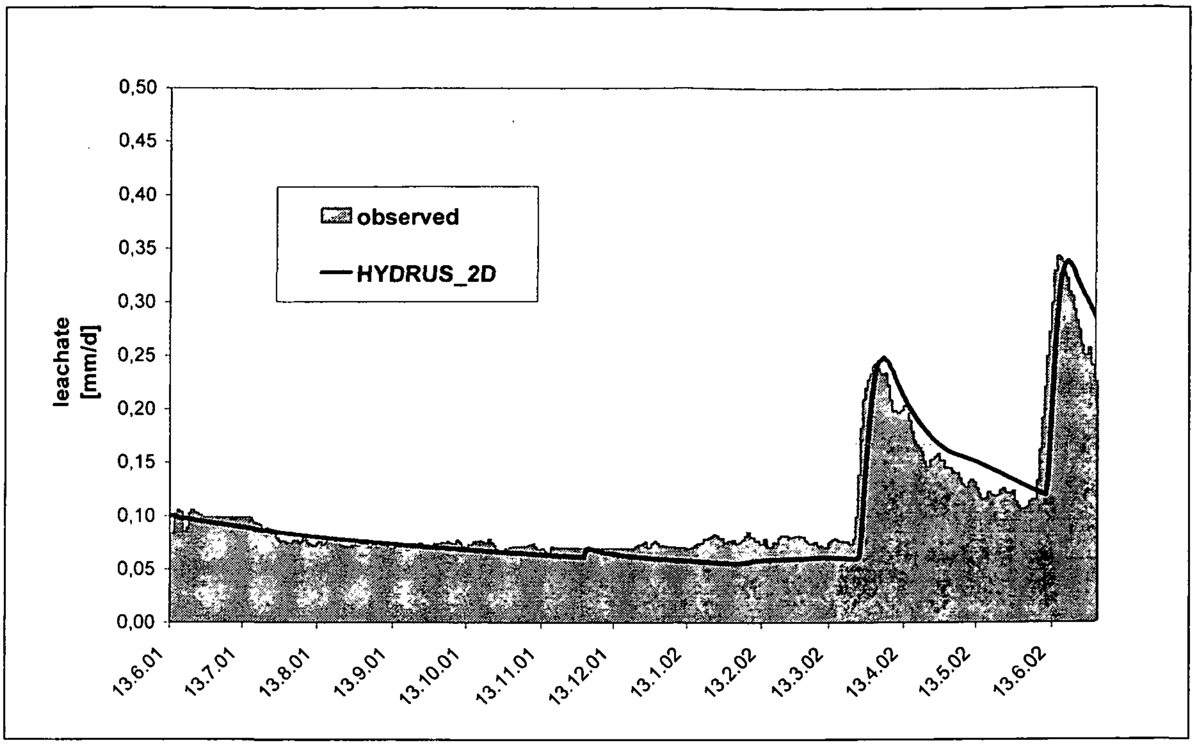



*Figure 5-11* ***Observed and simulated leachate discharge (Compartment II)***

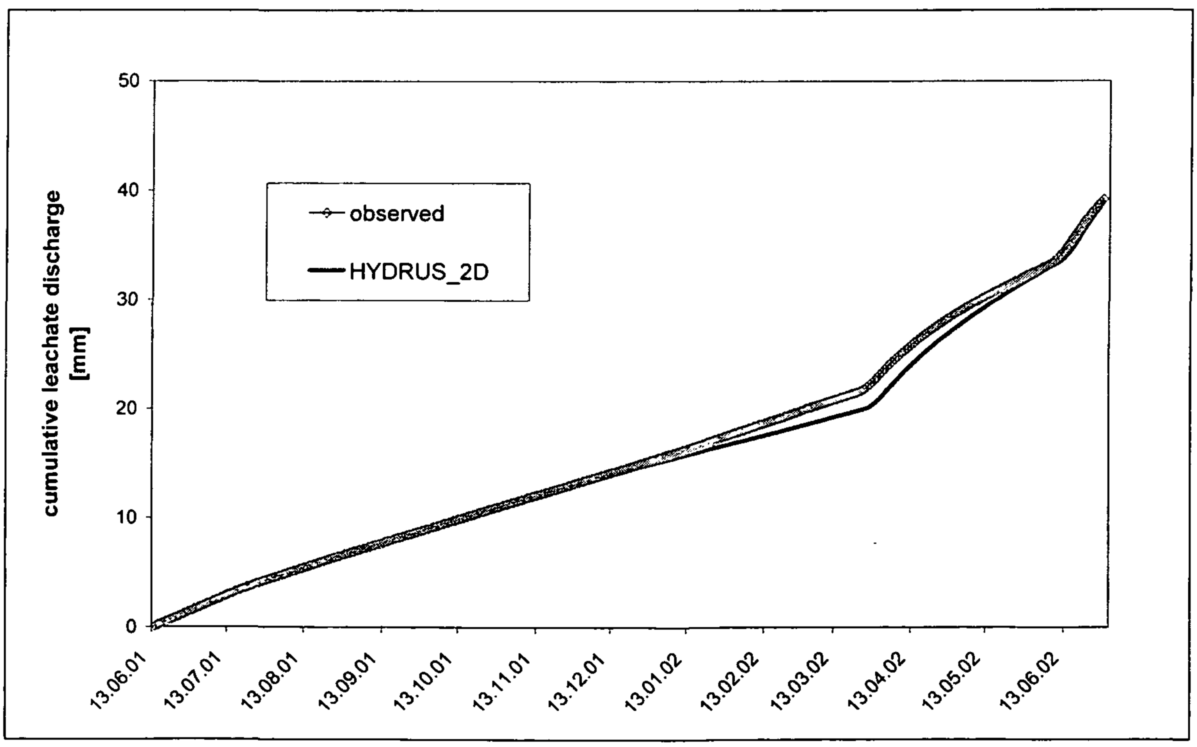



*Figure 5-12* ***Observed and simulated cumulative leachate discharge (Compartment II)***

### 5.1.5. Results of Compartment III

The leachate hydrograph observed at Compartment III is similar to that of Compartment II. No significant effect of rainfall events on discharge is observable for the time from June 2001 till July 2002. All precipitation evaporated due to the dry weather conditions and the capping system with a dense vegetation cover. Only close to the end of the observation period (June 2002) an increase of the leachate discharge caused by precipitation was noticed. Therefore, the recorded data set only enables to calibrate the water flow model. A following validation of the calibrated model would require a further time series that include changes in leachate generation rate. The maximum root depth representative for the vegetation cover of Compartment III was set analogous to C II/2 to 45 cm.

The calibration of the model results in the parameter values given in Table 5-9 and Table 5-10. These figures however, must be evaluated taking into account that almost no influence of rainfall on the leachate discharge was observable during the calibration period. Consequently, the performed calibration of the model is of low reliability.

***Table 5-9 Water retention parameters for Compartment III (Breitenau)***

| Material | Water content | | Retention | |
|---|---|---|---|---|
| | Residual $\theta_r$ [-] | Saturated $\theta_s$ [-] | Coefficient $\alpha$ [1/cm] | exponent $n_g$ [-] |
| Compost | 0.2 | 0.5 | 0.015 | 1.2 |
| Gravel (agglutinated) | 0.04 | 0.23 | 0.04 | 1.5 |
| Matrix-domain (waste body) | 0.15 | 0.5 | 0.02 | 1.4 |
| Preferential flow path | 0 | 0.01 | 0.02 | 1.4 |

***Table 5-10 Hydraulic conductivity parameters for Compartment III (Breitenau)***

| Material | Hydraulic conductivity | |
|---|---|---|
| | Saturated $K_s$ [cm/d] | Pore-connectivity l [-] |
| Compost | 300 | 0.5 |
| Gravel (agglutinated) | 7 | 0.5 |
| Matrix-domain (waste body) | 5 | 26 |
| Preferential flow path | 500 | 0.5 |

The ratio between horizontal and vertical conductivity (anisotropy) of the matrix domain was aligned to 0.3. Additionally, the scaling factor of the pressure head $\alpha_h$ for the channel domain was adjusted to 1.2 to achieve an agreement between simulated and observed leachate hydrographs.

Figure 5-13 shows predicted and measured discharge versus time. A good match between simulation results and observations was achieved. However, the capability of the calibrated model for predicting future leachate generation must be validated with another data set. Investigations within the scope of an ongoing research project "A New Method to Characterize the Stability of Old, Large Size landfills" (Döberl et al., 2004) will enable to evaluate the reliability of the calibrated water flow model at Compartment III.

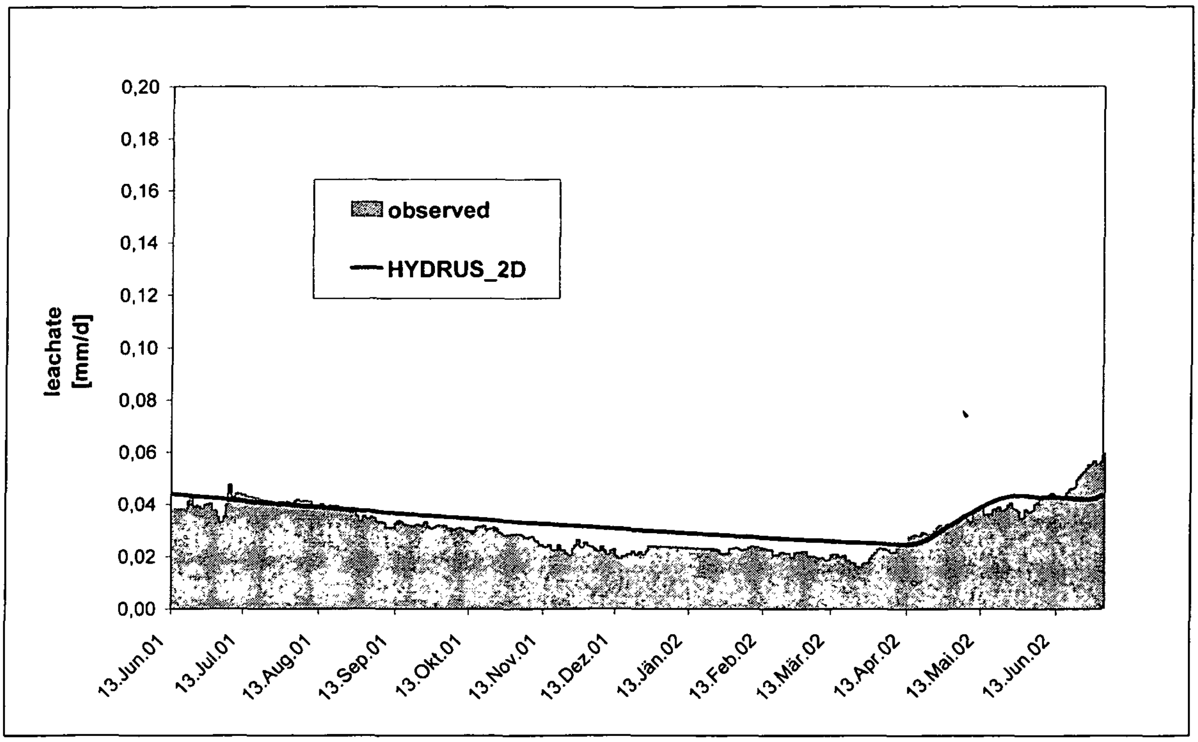



***Figure 5-13    Observed and simulated leachate discharge (Compartment III)***

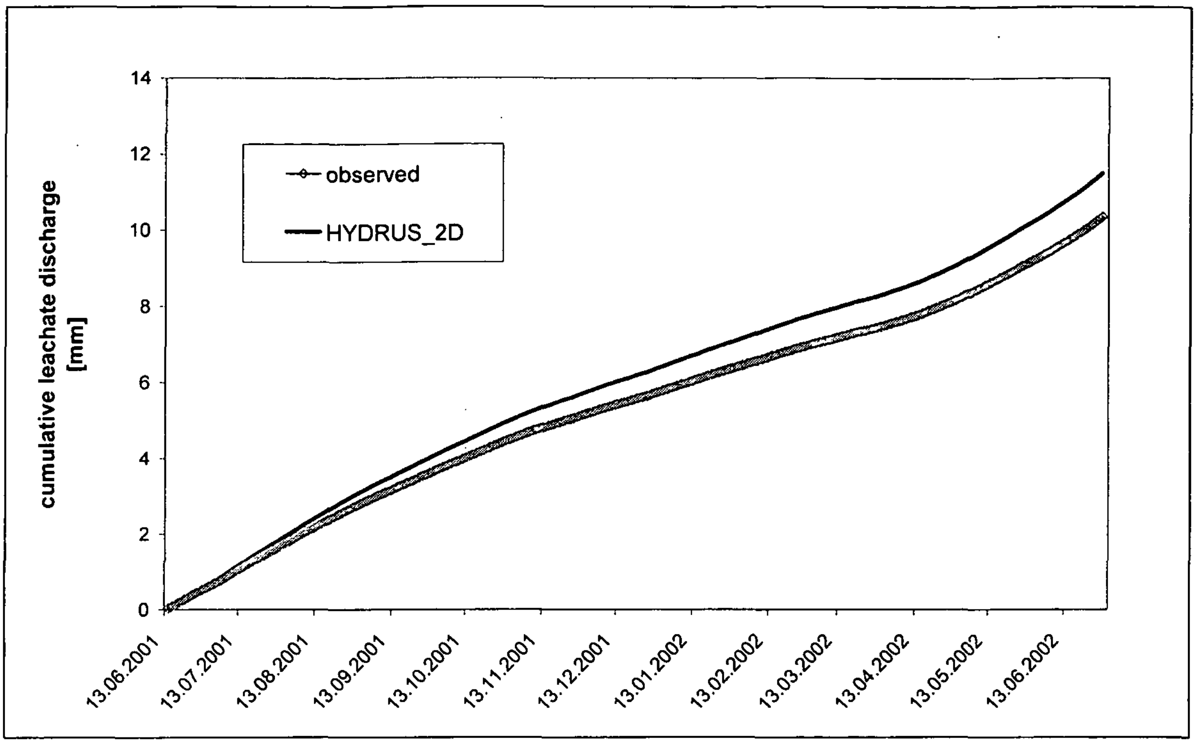



***Figure 5-14 Observed and simulated cumulative leachate discharge (Compartment III)***

## *5.2. Case study Spillepeng test cells*

### 5.2.1. Site description

The second case study was carried out using data from MSW test cells in Sweden. The considered landfills were constructed in 1988 at the Spillepeng landfill site in the city of Malmö.

The purpose of the original project (Nilsson et al., 1991) was to evaluate the dependence of biogas production on waste composition. Altogether six test cells, each with different waste composition as shown in Table 5-11, were constructed and operated over seven years, from 1989 till 1995. The volume of each cell is approximately 8,000 m³ and the cells contain about 4,000 tons of waste. The bottom dimensions are 35×35 m and the landfill surface is sloping, so that the height is decreasing from about 10 to 2 m (Figure 5-15).

The cells have been covered immediately after landfilling with 0.5 m clay. One year after closure in August/September 1990 an additional cover of 0.5 m plant soil was placed. Grass was sown in October 1990 and the vegetation has become established in the summer of 1991 (Nilsson et al., 1992).

***Table 5-11*** ***Waste composition and characteristics of the Spillepeng test cells (Nilsson et al., 1997)***

| | Waste composition | Volume [m³] | Mass [ton] | Average height [m] | Height (incl settlements) [m] | Wet density [kg/dm³] | Init. water content [m%] WS |
|---|---|---|---|---|---|---|---|
| **Cell 1** | Household - (70%) and industrial waste (30 %) | 7,400 | 3,400 | 6.0 | 5.7 | 0.48 | 23±5 |
| **Cell 2** | Household - (70%), industrial waste (30%) and sewage sludge (5%) | 6,800 | 3,200 | 5.6 | 5.3 | 0.53 | 26±5 |
| **Cell 3** | Household waste enriched with food waste fractions, horse manure | 7,600 | 3,500 | 6.2 | 5.9 | 0.48 | 20±5 |
| **Cell 4** | Household waste (100%) | 8,000 | 5,200 | 6.5 | 6.2 | 0.68 | 36±5 |
| **Cell 5** | Household waste (100%) and sewage sludge (5%) | 8,400 | 4,800 | 6.9 | 6.5 | 0.62 | 34±5 |
| **Cell 6** | Household waste (100%) | 7,600 | 5,200 | 6.2 | 5.9 | 0.72 | 33±5 |

The bottom liner of the cells consists of 0.5 m clay and a geomembrane liner. For the protection of the plastic liner, sand layers of 20 cm were placed below and above the geomembrane. The cross section of the cells is shown in Figure 5-15.

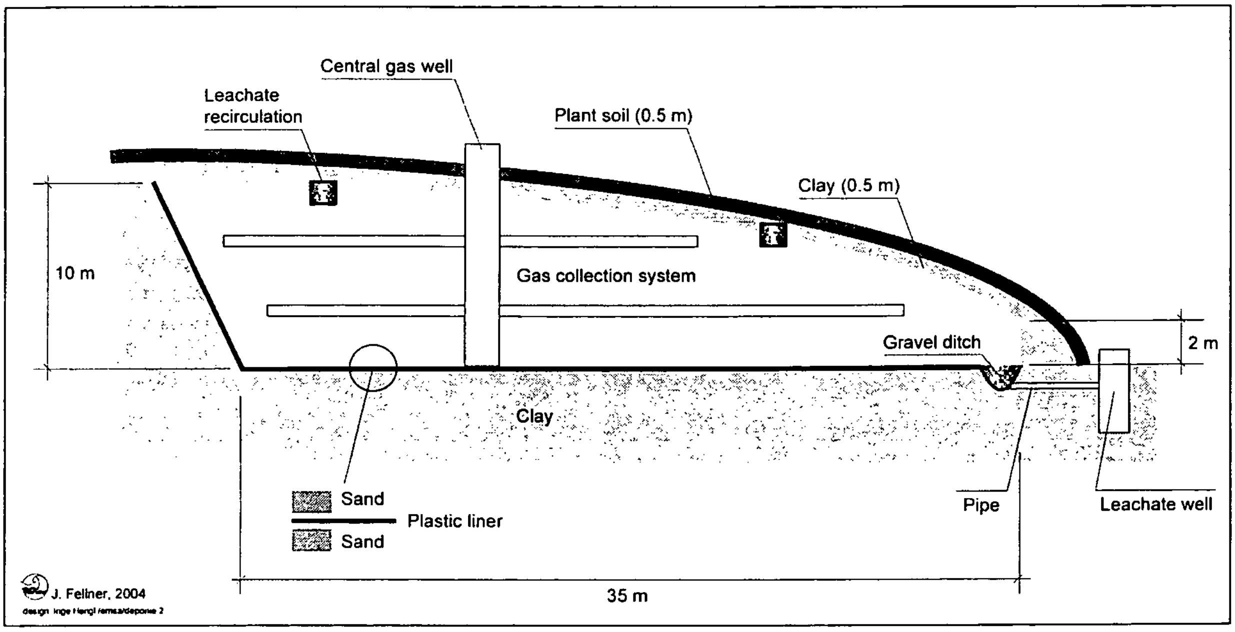



*Figure 5-15 Spillepeng test cell construction – cross section (Åkesson & Nilsson, 1997)*

Leachate is collected at the lower end of the cell, where a gravel ditch has a lined connection (PVC-pipe) to a leachate well of 2 m³. The quantity of generated leachate was determined by measuring the water level in the leachate tank. The tank was emptied manually before it became full. At high flow rates, the water level in the tank could exceed the level of the pipe connecting the well to the gravel ditch inside the test cell, thereby preventing the leachate from draining. Consequently, additional discharge was draining when the well was emptied. These extra volumes were quantified through the time of pumping. In the case that the well was not emptied regularly a larger amount of leachate was retained inside the test cells (within the gravel ditch) and it was impossible to get information on the temporal variation in the collected leachate. To avoid retention of water inside the cells (in particular inside the gravel ditch) during periods with high rate of leachate generation, the plant was upgraded in December 1994, so that the wells were emptied automatically. Since this date the recorded pumping time was used to determine the leachate discharge.

For the observation period from January 1989 to December 1995 the recorded data of the leachate amount was available on a weekly basis. Some periods however had a lower temporal resolution. In particular measurements were missing for the time between July 1990 and January 1991. Furthermore, the emptying of leachate wells was disregarded in the period from February till August 1994. Well documented data with high temporal resolution exist for the last year of observation (1995), after the upgrading of the leachate management system.

### 5.2.2. Input information

#### 5.2.2.1. Meteorological data (Spillepeng)

The meteorological data such as precipitation, temperature, humidity, wind speed and sunshine hours (necessary to calculate the potential evapotranspiration), were obtained from the nearby meteorological stations Malmö (3 kilometres distance) and Lund (around 15 kilometres). In particular values for precipitation, temperature and humidity were taken from Malmö, whereas information on sunshine hours was only available from the station in Lund. During periods with malfunctioning of the meteorological station in Malmö, the data from Lund had to be consulted.

The reference evapotranspiration $ET_0$ for the Spillepeng site was estimated using the method after Penman-Moneith (Bevan, 1979). In order to obtain the potential evapotranspiration $ET_p$ of the considered vegetation the reference values $ET_0$ were multiplied by a constant crop coefficient $K_c$ of 1.15 (dense vegetation cover) for the growing period (April till October). Beyond this period a dependence of $K_c$ from the reference evapotranspiration $ET_0$ according to Allen et al. (1998) was used (see Figure 5-4).

Surface runoff from the landfill cover was only measured for short time (from October 1993 till March 1994) and assumed to be negligible (Bendz et al., 1997). However, the distinct slope (15 %) of the landfill surface calls for considering overland flow. In particular short time after landfill closure (no vegetation cover) a considerable amount of runoff was observed by the operational staff of the Spillepeng site (Andersson, 2003). As measured data were not available, runoff was calculated according to the common SCS-curve number method (Soil Conservation Service, 1973). This approach accounts for the vegetation cover, the water content of the cover layers and the slope of the surface.
The results of the calculations show that with increasing time after landfill closure and therefore denser crop cover runoff is declining (Figure 5-16).

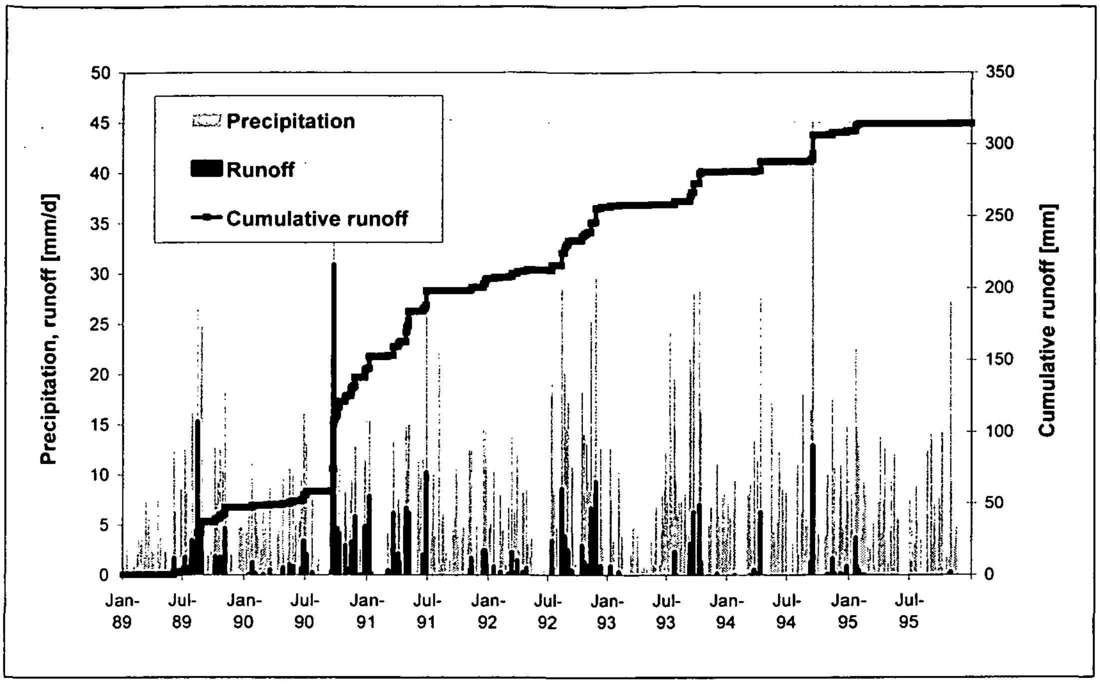



*Figure 5-16* ***Precipitation and estimated runoff (Spillepeng test cells)***

### 5.2.2.2. Hydraulic properties

The required hydraulic characteristics of the cover materials were determined by field and laboratory experiments performed by the author during a research visit at Lund University (lasting from January till June 2003). The hydraulic conductivity was measured using infiltration tests. So-called inversed bore-hole tests (Klute, 1986) were performed. The porosity, the bulk density and the retention characteristics (curve) were determined in the laboratory using undisturbed soil samples. The pressure cell method after Richards (1941) was applied to derive the relation (retention characteristics) between matrix potential (pressure head) and water content of the two cover soils. In Table 5-12 and Table 5-13 the results of the hydraulic investigations are presented.

***Table 5-12*** ***Conductivity, porosity and density of the landfill cover layers (Spillepeng cells)***

| Cover material | Saturated hydraulic conductivity $K_s$ [cm/d] | Porosity n [-] | Bulk density $\rho_d$ [kg/dm³] |
|---|---|---|---|
| „Plant soil" | 20 – 120 | 0.33 – 0.41 | 1.59 – 1.75 |
| „Clay" | 0.6 – 8 | 0.34 – 0.43 | 1.52 – 1.80 |

*Table 5-13 Estimated van Genuchten parameters of the landfill cover layers (Spillepeng cells)*

| Cover material | Methods | van Genuchten parameter | | | |
|---|---|---|---|---|---|
| | | Residual water content $\theta_r$ [-] | Saturated water content $\theta_s$ [-] | Parameter $\alpha$ [1/cm] | Parameter $n_g$ [-] |
| „Plant soil“ | Pressure cell, curve fitting | 0.04 – 0.09 | 0.29 – 0.35 | 0.042 – 0.081 | 1.44 – 1.66 |
| „Clay“ | Pressure cell, curve fitting | 0.07 – 0.15 | 0.34 – 0.39 | 0.010 – 0.023 | 1.25 – 1.39 |

First estimates concerning the hydraulic characteristics of the waste "domains" are corresponding to those for the landfill Breitenau. Also the same calibration procedure (trial and error method with graphical evaluation) was used (e.g. see chapter 5.1.3). Due to the minor leachate discharge during the first years and the poor temporal resolution of available data during this time, the calibration was performed using the time series from September 1994 till December 1995. This period coincides with the phase of automatic emptying of the leachate collection wells. Additionally the total cumulative amount of generated leachate during the whole observation period (1989-1995) was used for calibration purposes.
The introduced water flow model was calibrated at Cell 1, Cell 2 and Cell 4 only. For the other cells either reasonable suspicions that the leachate collection system was malfunctioning exist (Cell 3 and Cell 5, Bendz et al., 1997) or the documentation about operational data was insufficient (leachate recirculation at Cell 6).

### 5.2.3. Results of Cell 1

Analogous to the water flow modeling at the Breitenau landfill, separate simulations were carried out for the landfill cover and the waste body. The results of the simulations for the landfill cover (seepage output of HYDRUS-2D) served as input information for the hydrologic system waste body that consists of the matrix domain and the channel domain. The calculations for this system were performed for an average landfill profile (height for Cell 1 equals 5.7 m) with a width of 0.5 m (see Figure 5-17). The reduction of the profile width compared to the simulations for the landfill Breitenau is necessary to get parameter values that are comparable to those obtained for landfills of bigger height. The root depth of the vegetation was set according to field investigations to 30 cm, whereby in order to simplify matters a constant distribution of roots over the depth was assumed. The initial mass water content of the landfilled waste in Cell 1 was reportedly around 23 % (referred to wet mass). This results for a given waste density of 0.48 kg/dm³ in a volumetric water content θ of 0.11. This value was used as initial condition for the water flow simulations.

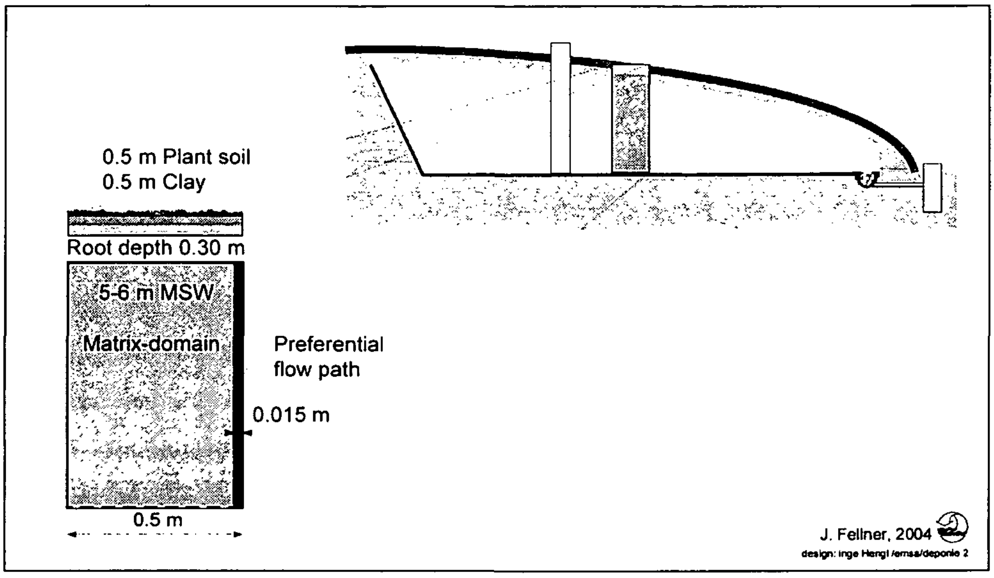



*Figure 5-17 Simplified model system (Spillepeng test cells)*

Best match between predicted and observed leachate discharge was achieved using the parameter values of the landfill cover layers and the waste domains given in Table 5-14 and Table 5-15.

*Table 5-14 Water retention parameters for Cell 1 (Spillepeng landfill)*

| Material | Water content | | Retention | |
|---|---|---|---|---|
| | Residual $\theta_r$ [-] | saturated $\theta_s$ [-] | Coefficient $\alpha$ [1/cm] | Exponent $n_g$ [-] |
| Plant soil | 0.06 | 0.31 | 0.05 | 1.53 |
| Clay | 0.1 | 0.35 | 0.014 | 1.26 |
| Matrix-domain (waste body) | 0.02 | 0.42 | 0.02 | 1.4 |
| Channel domain | 0 | 0.01 | 0.02 | 1.4 |

*Table 5-15* ***Hydraulic conductivity parameters for Cell 1 (Spillepeng landfill)***

| Material | Hydraulic conductivity | |
|---|---|---|
| | Saturated $K_s$ [cm/d] | Pore-connectivity l [-] |
| Plant soil | 50 | 0.5 |
| Clay | 5 | 0.5 |
| Matrix-domain (waste body) | 3 | 11 |
| Channel domain | 300 | 0.5 |

Additionally to the above listed parameter values, the anisotropy of the matrix domain concerning the hydraulic conductivity $K^A_h$ (representing the ratio between horizontal and vertical hydraulic conductivity $K_h/K_v$) was set to 0.45 and the scaling factor for the pressure head $\alpha_h$ of the preferential flow path to a value of 2 for best agreement between simulated and measured discharge during the calibration period (see Figure 5-18).

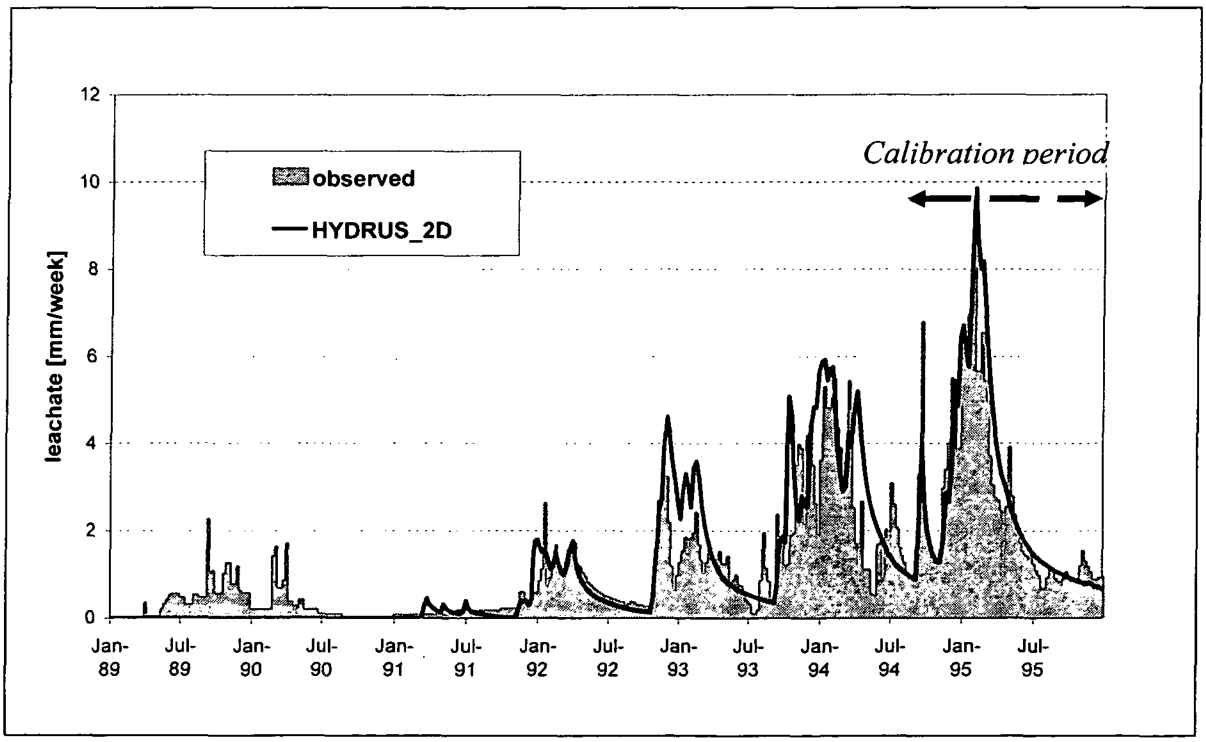



***Figure 5-18*** ***Observed and simulated leachate discharge (Cell 1)***

The validation of the model shows that also beyond the calibration time, the discharge was predicted quite accurately. Only during the first years after landfilling (1989-1990) no leachate outflow was simulated, which is contrary to the observation. This fact is attributed to the simplification of the test cell with different heights to an average profile with a constant depth of 5.7 m, thereby neglecting areas within the cell of 2 m depth only. Water storage capacity within this lower end of the test cell is exceeded faster than the capacity of the modeled (simplified) landfill profile would admit. Thus, first discharge from the test cell occurs earlier compared to a landfill of constant depth (as assumed for the modeling effort). Furthermore, differences between measured and predicted leachate generation can be attributed to uncertainties associated with the evaluation of runoff and evapotranspiration. Some deviations (e.g. spring till summer 1994) however, refer to discontinuous emptying of the leachate collection tank, and thus, misleading observed leachate discharge. Partly misrepresented observation data is also the reason for renouncing the determination of a quality grade which evaluates the match between simulated and measured leachate generation rate.

Observed and predicted cumulative discharge (Figure 5-19) show small differences, that attribute as mentioned above on the one hand to simplifications of the landfill geometry and on the other hand to uncertainties associated with the water input into the landfill. Water storage processes within the landfill body seem to be described adequately as calculated and observed water content at the end of the simulation period are corresponding (Figure 5-24) well.

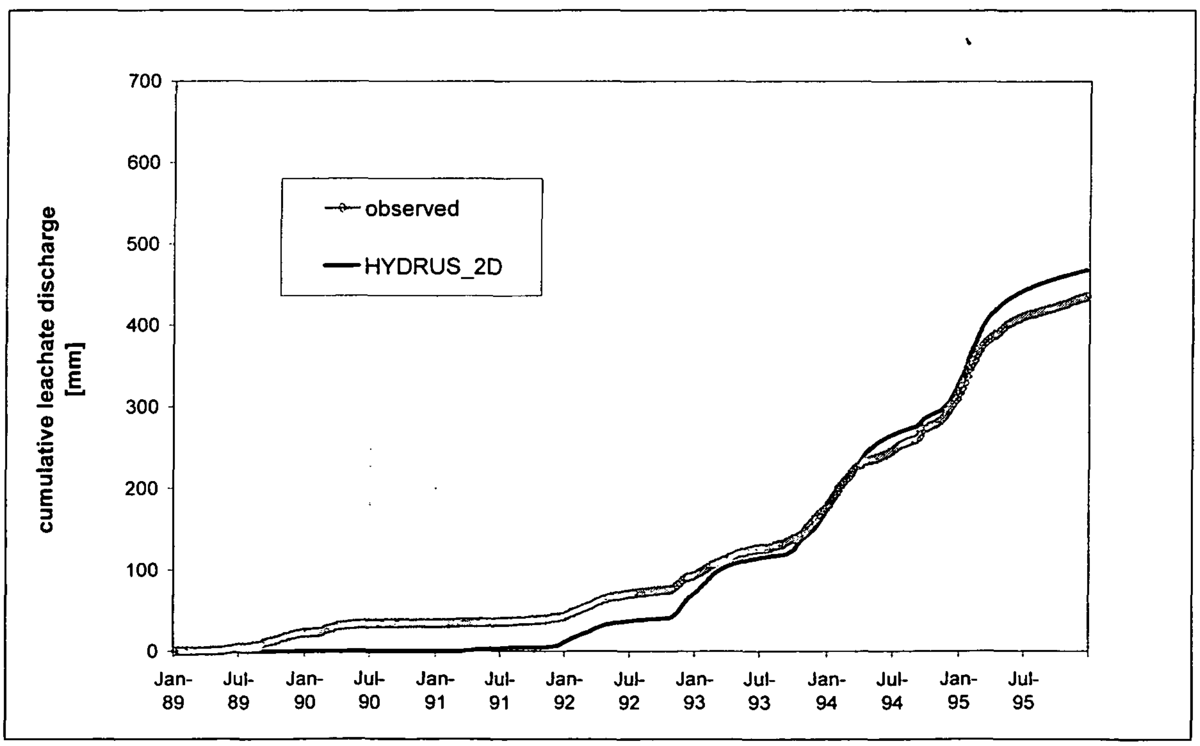



***Figure 5-19 Observed and simulated cumulative leachate discharge (Cell 1)***

### 5.2.4. Results of Cell 2

The leachate hydrograph at Cell 2 shows a similar shape to those at Cell 1. This is due to the same composition of the test cells (size, cover layers). The rate and cumulative amount of water drained from Cell 2 however, is higher compared to the discharge of Cell 1. This may be due to the different initial water content (Table 5-11), and thus, different available water storage capacities. The fact that the average landfill height of both test cells is somewhat different may also affect the water storage and the leachate generation. According to reported data the simulations were carried out with an initial volumetric water content θ of the waste matrix of 0.14.

The calibration of the water flow model was limited to hydraulic characteristics of the waste domains only, as parameters of the cover layers had already been determined for Cell 1. The results of the calibration are presented in Table 5-16 and Table 5-17.

*Table 5-16 Water retention parameters for Cell 2 (Spillepeng landfill)*

| Material | Water content | | Retention | |
|---|---|---|---|---|
| | Residual $\theta_r$ [-] | Saturated $\theta_s$ [-] | Coefficient $\alpha$ [1/cm] | Exponent $n_g$ [-] |
| Matrix-domain (waste body) | 0.02 | 0.42 | 0.02 | 1.4 |
| Channel domain | 0 | 0.01 | 0.02 | 1.4 |

*Table 5-17 Hydraulic conductivity parameters for Cell 2 (Spillepeng landfill)*

| Material | Hydraulic conductivity | |
|---|---|---|
| | Saturated $K_s$ [cm/d] | Pore-connectivity $l$ [-] |
| Matrix-domain (waste body) | 3 | 9 |
| Channel domain | 600 | 0.5 |

Additionally the calibration of the water flow model leads for the matrix domain to an anisotropy of the hydraulic conductivity $K^A_h$ of 0.6, and for the channel domain to a pressure head scaling factor $\alpha_h$ of 3.

The simulation results (Figure 5-20) indicate that the model predicts leachate generation quite accurately even beyond the calibration period. Analogously to simulations for Cell 1, deviations are noticeable only short time after landfilling as well as during periods when leachate collection wells were emptied erratically. The largest errors between measured and predicted cumulative discharge (Figure 5-21) occurred during the second year after landfill closure, when nearly no drainage was predicted by the model. During the remaining time simulated, the maximum and average errors were 30 % and 8 %, respectively.

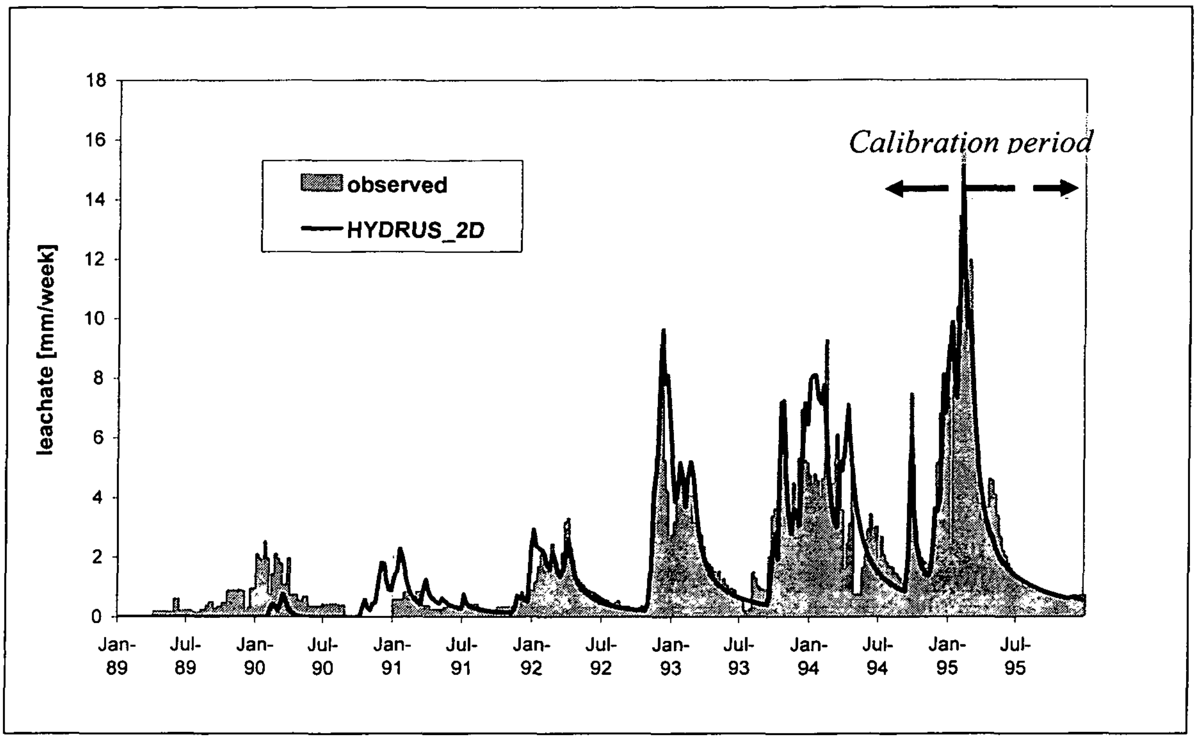



***Figure 5-20 Observed and simulated leachate discharge (Cell 2)***

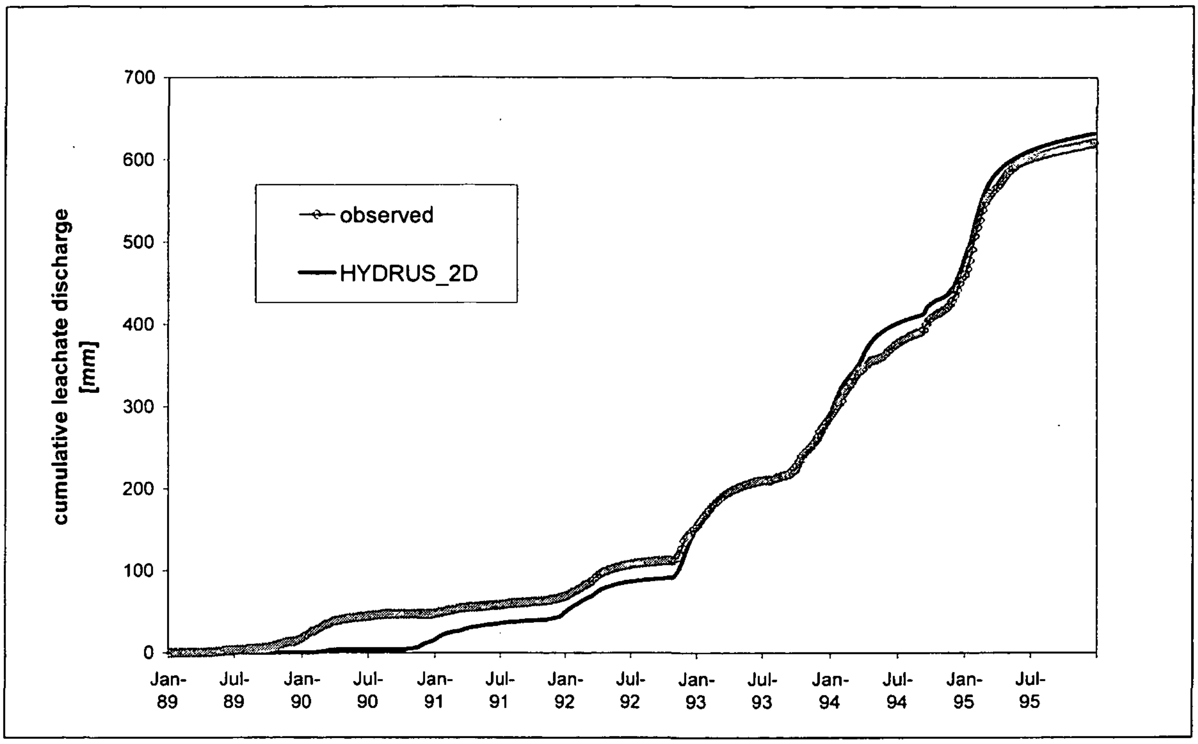



*Figure 5-21* ***Observed and simulated cumulative leachate discharge (Cell 2)***

### 5.2.5. Results of Cell 4

The average landfill height of Cell 4 is around 6.2 m. The landfilled waste shows an initial mass water content of 36 % (referred to wet mass), that corresponds to a volumetric water content of 0.25. Compared to Cell 1 and 2 the water content of the waste is higher which probably attributes to different composition (Nilsson et al., 1997). The self-evident assumption that soggier waste will generate more leachate was not confirmed by the observation. Moreover Cell 4 shows least leachate generation during the period from 1989 till 1995 (Cell 1 ~ 430 mm, Cell 2 ~ 615 mm, Cell 4 ~ 340 mm). This may be due to a higher water sorption capability of the waste landfilled or because of a more uniform water distribution within Cell 4. Both facts are associated with a larger effective water storage capacity of the landfill. The calibrated parameter values of the water flow model can provide an indication of the predominating process responsible for enhanced water retention within Cell 4 (Figure 5-24).

In order to reach best match between predicted and observed leachate discharge, model parameters had to be adjusted to the values given in Table 5-18 and Table 5-19. Furthermore, anisotropy of the hydraulic conductivity $K^A_h$ had to be set to 0.2 and the pressure head scaling factor $\alpha_h$ for the channel domain to 1.8.

*Table 5-18* ***Water retention parameters for Cell 4 (Spillepeng landfill)***

| Material | Water content | | Retention | |
|---|---|---|---|---|
| | Residual $\theta_r$ [-] | Saturated $\theta_s$ [-] | Coefficient $\alpha$ [1/cm] | Exponent $n_g$ [-] |
| Matrix-domain (waste body) | 0.1 | 0.5 | 0.02 | 1.4 |
| Channel domain | 0 | 0.01 | 0.02 | 1.4 |

*Table 5-19* ***Hydraulic conductivity parameters for Cell 4 (Spillepeng landfill)***

| Material | Hydraulic conductivity | |
|---|---|---|
| | Saturated $K_s$ [cm/d] | Pore-connectivity l [-] |
| Matrix-domain (waste body) | 3 | 12 |
| Channel domain | 200 | 0.5 |

Figure 5-22 represents predicted and measured leachate generation as a function of time. Predicted and observed values are close excluding the first time after landfill closure. The same phenomenon was noticed for Cell 1 and 2, and is attributed to the simplification of the landfill geometry.

Comparing the calibrated model parameters of Cell 4 with those of Cell 1 and 2 indicates that the water sorption capability, represented somehow by the retention characteristics of the matrix domain ($\theta_s$, $\theta_r$, $\alpha$, n), was enhanced. The uniformity of the flow regime, expressed by smaller values of the hydraulic anisotropy $K^A_h$, the scaling factor for the pressure head $\alpha_h$ and the saturated hydraulic conductivity $K_s$ of the channel domain was also enhanced. Therefore higher water storage within Cell 4 can be ascribed to both reasons: higher water sorption capability of the waste mass itself and more uniform water flow.

Predicted and measured cumulative leachate discharge parallel each other closely (Figure 5-23). Differences result only from the first years after landfilling, when the model underpredicts leachate generation. The reason therefore is once again the simplification of the landfill geometry to a rectangular profile.

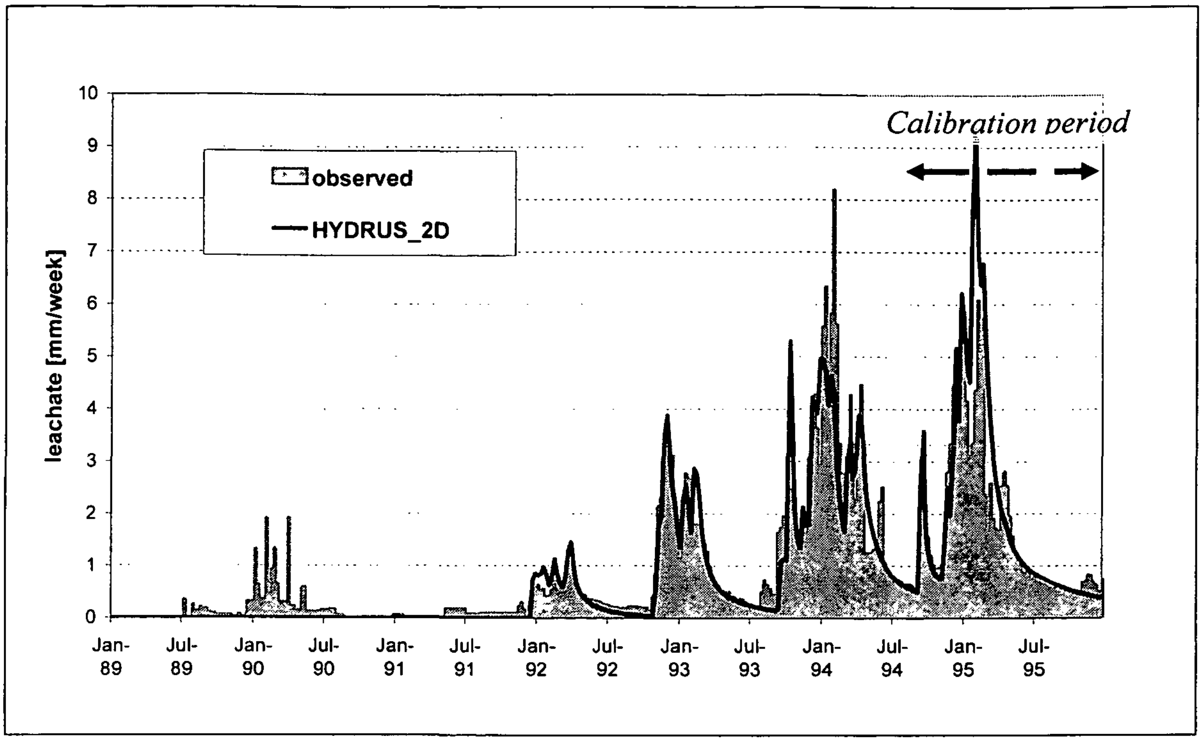



*Figure 5-22    Observed and simulated leachate discharge (Cell 4)*

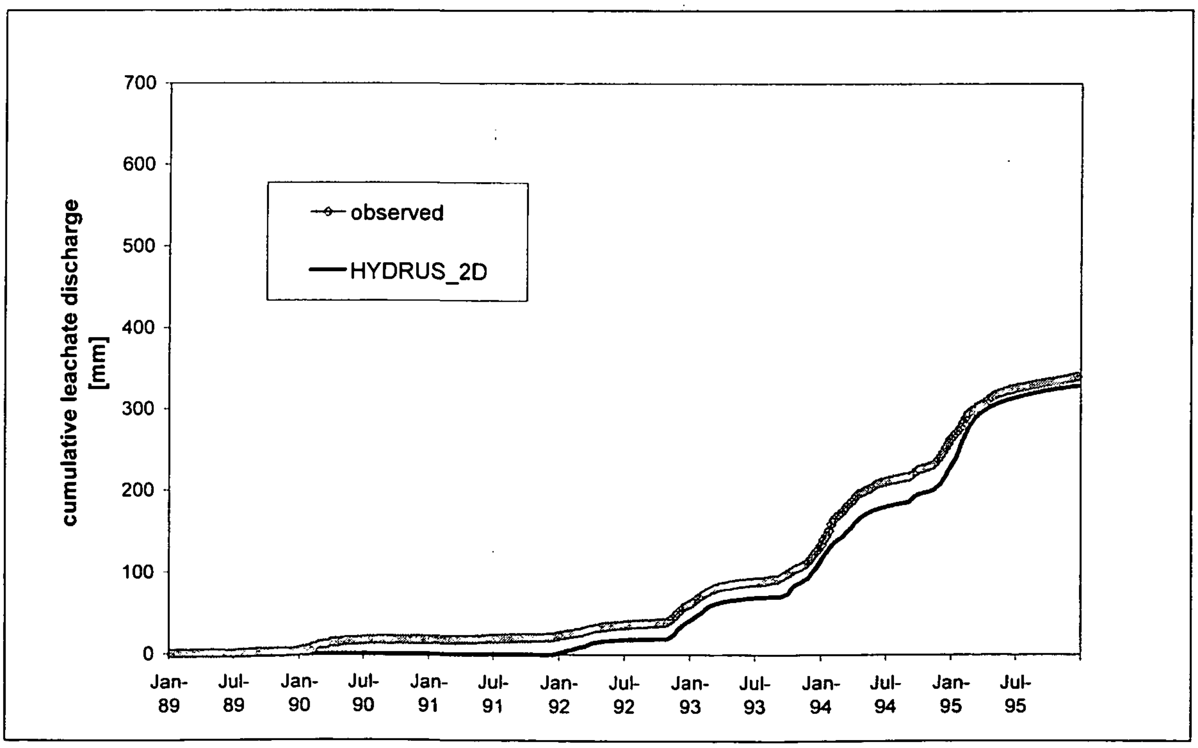



*Figure 5-23    Observed and simulated cumulative leachate discharge (Cell 4)*

Figure 5-24 shows predicted and observed average water content of all three cells at the end of the simulation period (December 1995). The results indicate that the model also accurately reproduces water storage processes within the landfill.

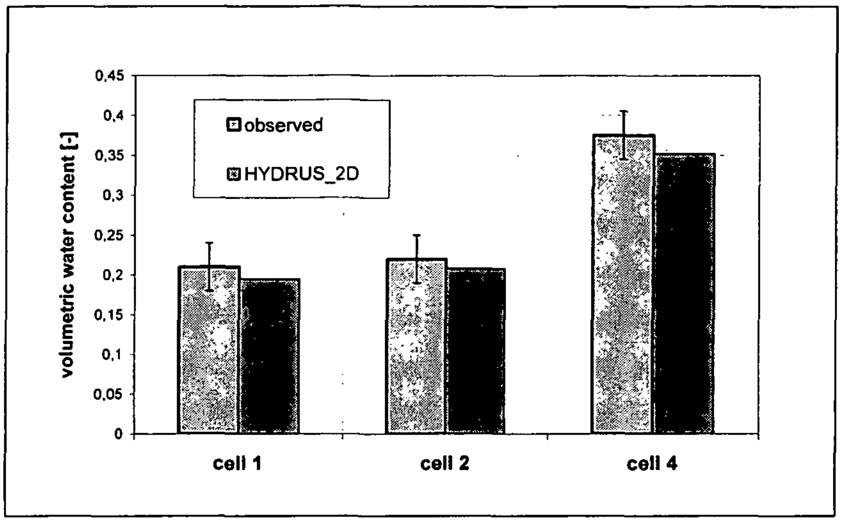
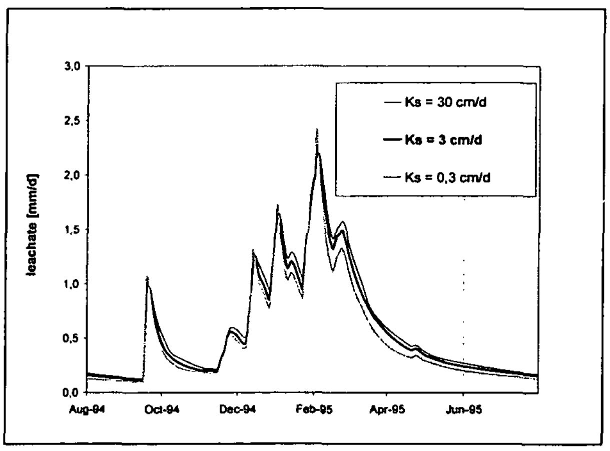



*Figure 5-24 Observed (Nilsson et al., 1997) and simulated average water content (Dec. 1995)*

## *5.3. Sensitivity analysis*

A set of simulation runs was performed to assess the model's sensitivity to its hydraulic parameters. The sensitivity analyses focused on parameters of the waste mass (matrix and channel domain). Hydraulic parameters of the cover layers and meteorological input data have not been investigated. Within the scope of the sensitivity analyses the parameter values obtained from the calibration study of the Spillepeng test Cell 2 were taken as standard values. Each of the parameters was varied within feasible ranges (see Table 4-5), while all the other parameters were kept constant.

Changes in the shape of the discharge hydrographs (e.g. ratio between base flow and discharge peaks) as well as in the cumulative leachate discharge due to parameter variation have been investigated. Some results are presented in the following figures, whereby only a period of one year (August 1994 till July 1995) is shown in order to facilitate the differentiation of the curves. The black thick line in the following figures represents the results for the calibrated parameter set.



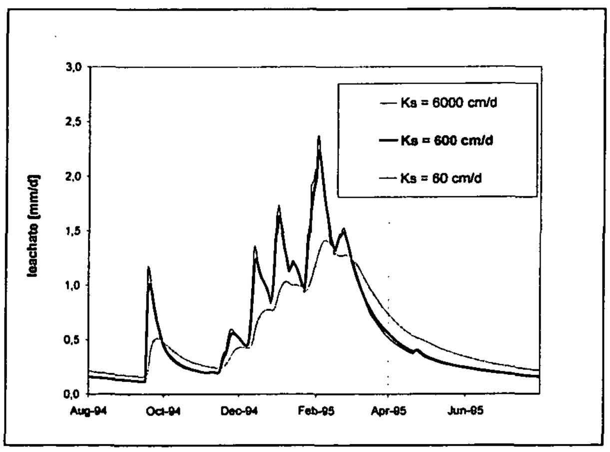




***Figure 5-25*** ***Sensitivity analysis for the saturated hydraulic conductivity $K_s$ of the matrix domain (left side) and the channel domain (right side) for Cell 2***

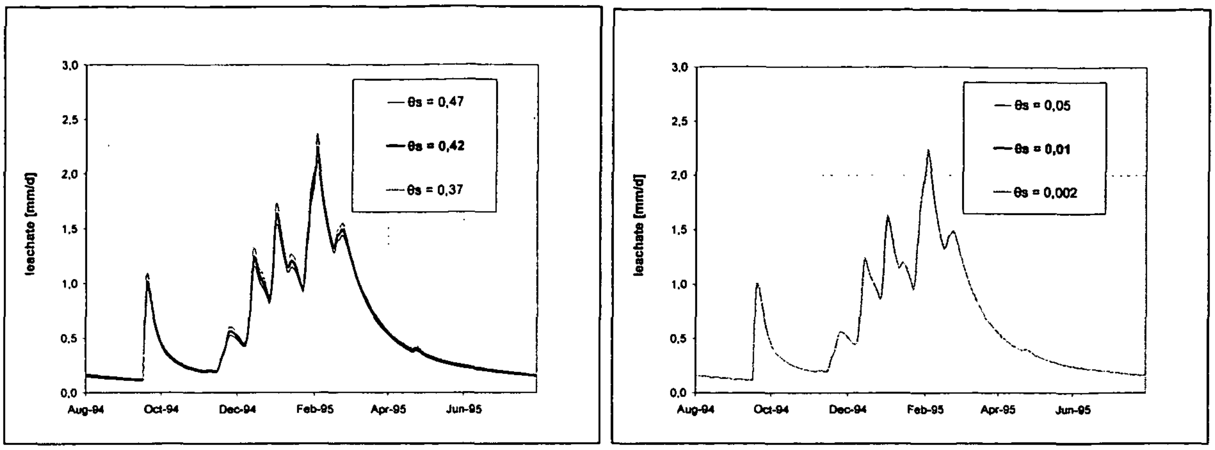



*Figure 5-26* **Sensitivity analysis for the saturated water content $\theta_s$ of the matrix domain (left side) and the channel domain (right side) for Cell 2**

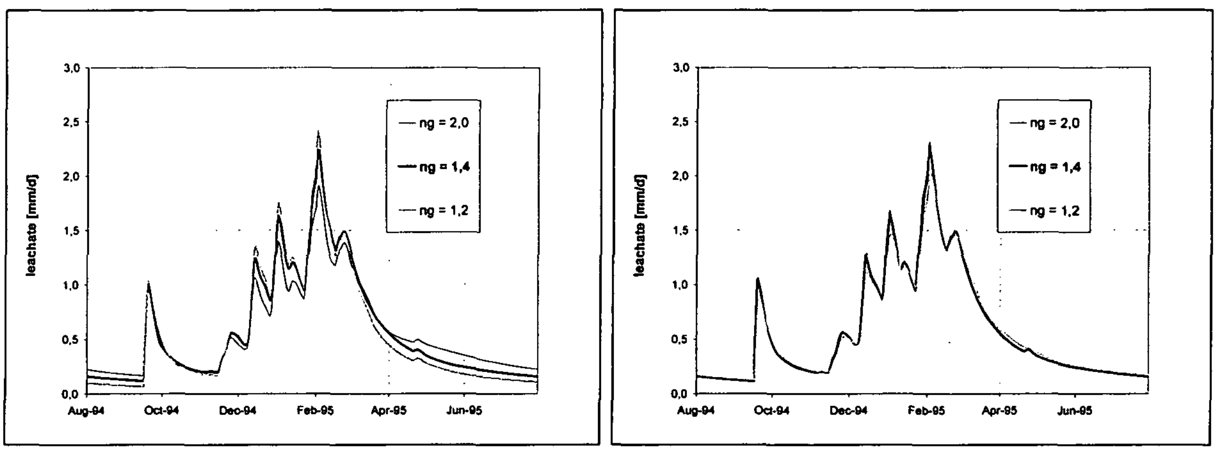



*Figure 5-27* **Sensitivity analysis for the form parameter $n_g$ of the matrix domain (left side) and the channel domain (right side) for Cell 2**

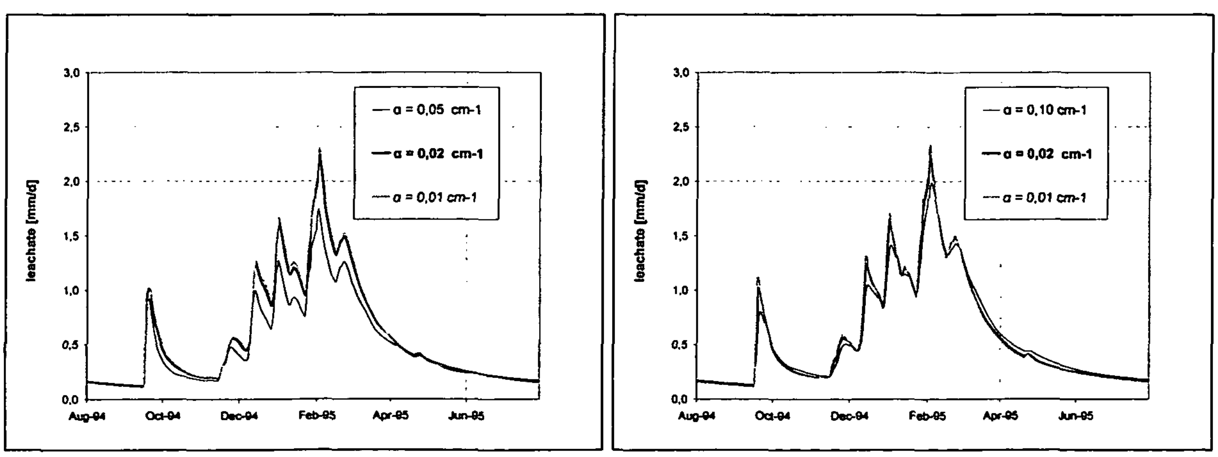
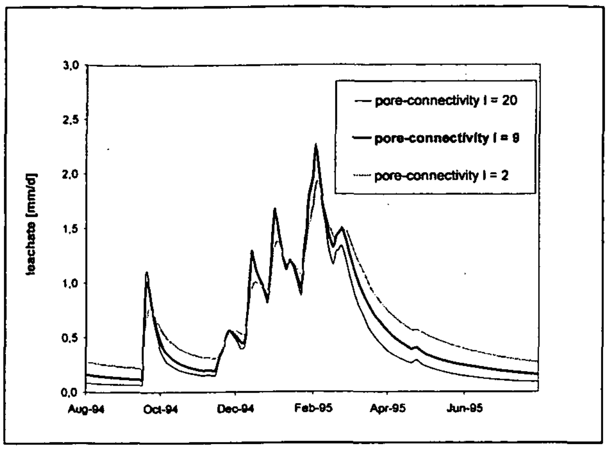



*Figure 5-28* **Sensitivity analysis for the form parameter $\alpha$ of the matrix domain (left side) and the channel domain (right side) for Cell 2**



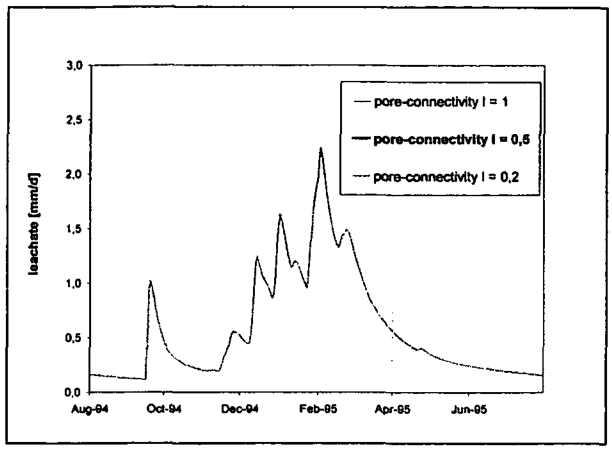




*Figure 5-29* ***Sensitivity analysis for the pore-connectivity parameter l of the matrix domain (left side) and the channel domain (right side) for Cell 2***

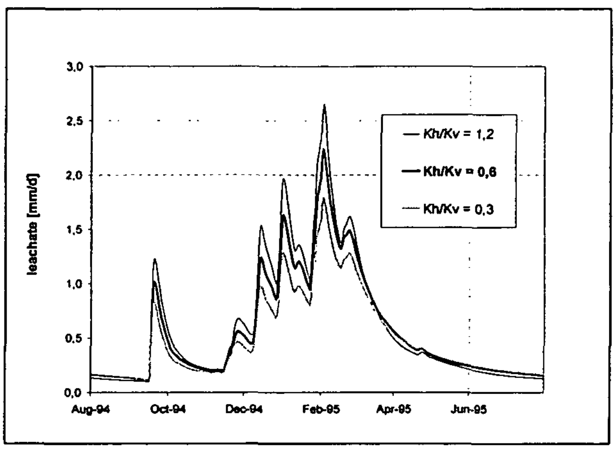



*Figure 5-30* ***Sensitivity analysis for the parameter hydraulic anisotropy $K_h/K_v$ of the matrix domain for Cell 2***

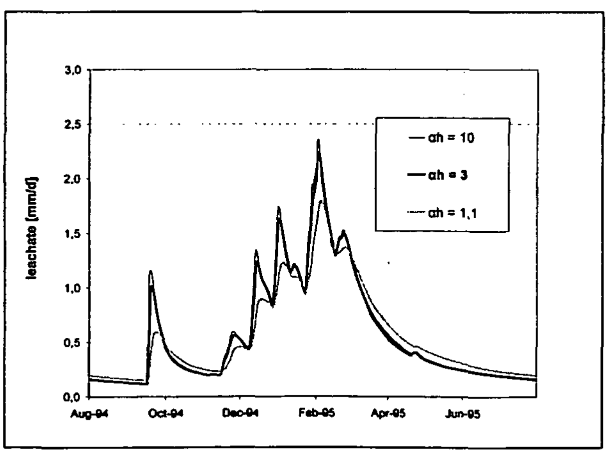



*Figure 5-31* ***Sensitivity analysis for the parameter pressure head scaling factor $\alpha_h$ of the channel domain for Cell 2***

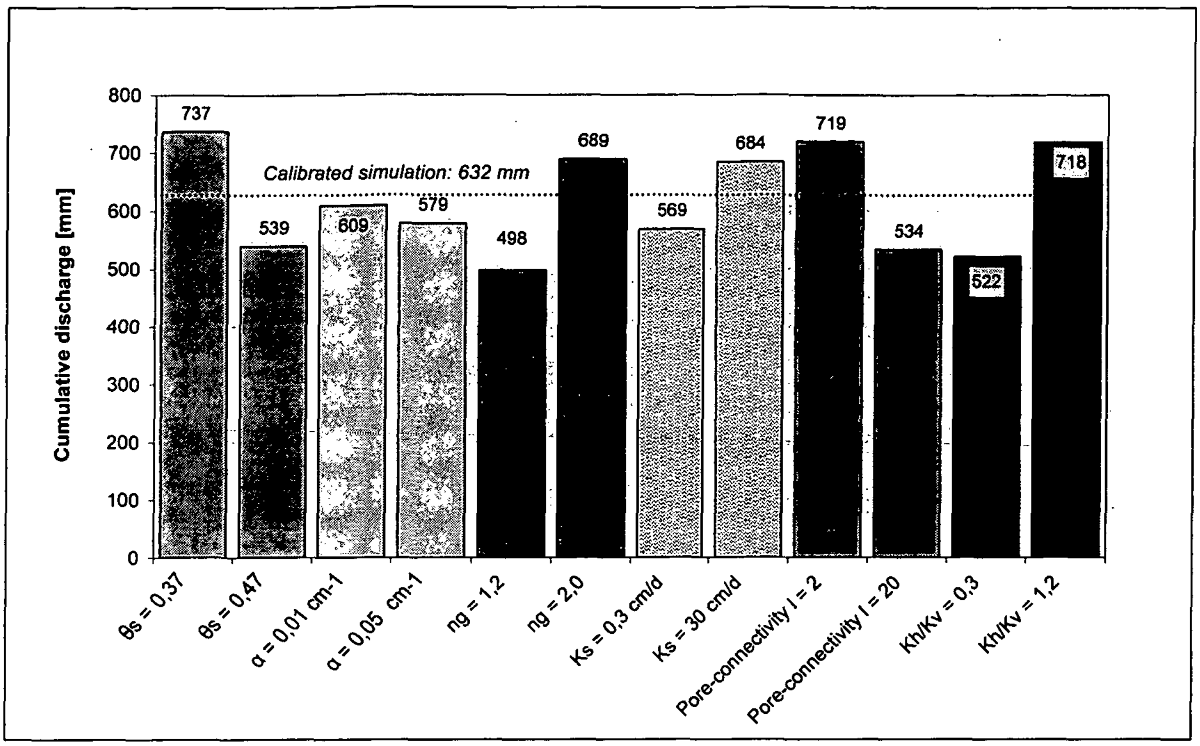



*Figure 5-32* ***Sensitivity analysis for model parameters of the matrix domain (cumulative discharge of Cell 2)***

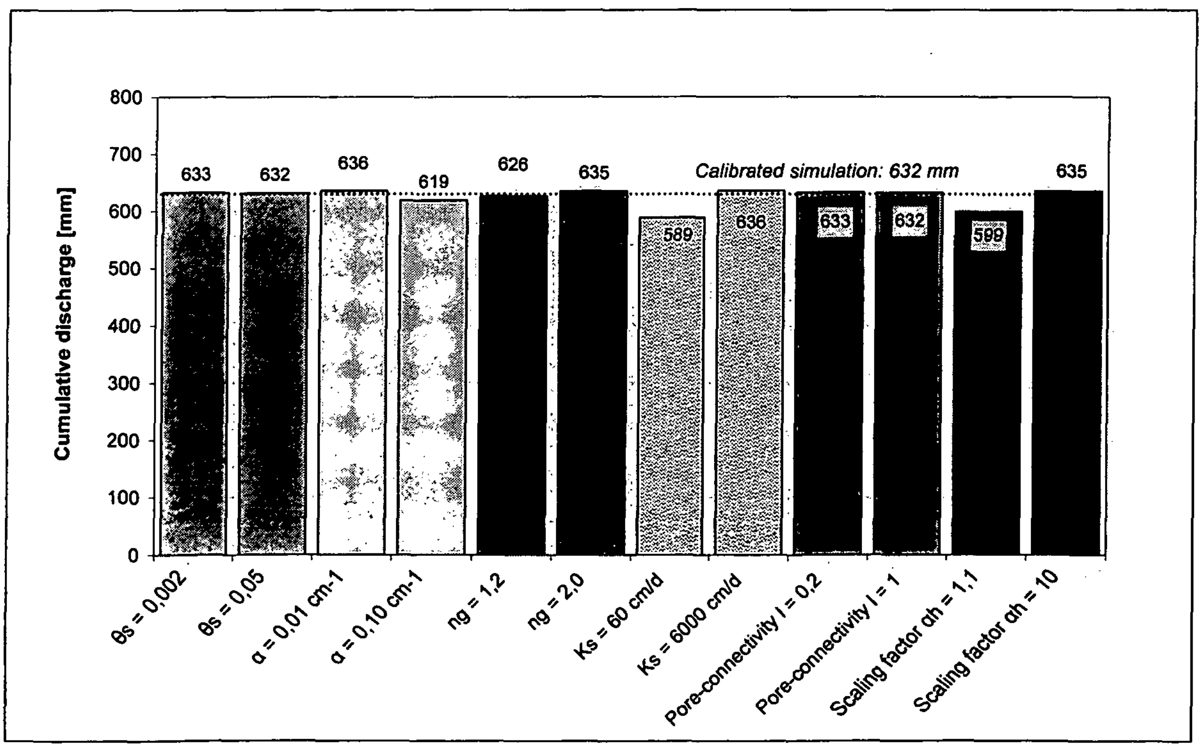



*Figure 5-33* ***Sensitivity analysis for model parameters of the channel domain (cumulative discharge of Cell 2)***

The sensitivity analyses showed that the results of the water flow model are mainly dependent on the hydraulic characteristics of the matrix domain. The variation of channel domain parameters had less impact on the results, with the exception of the saturated hydraulic conductivity $K_s$ and the pressure head scaling factor $\alpha_h$.

Total water storage, and thus cumulative leachate discharge is strongly affected by the following matrix domain parameters:

- residual $\theta_r$ and saturated water content $\theta_s$, respectively their difference,
- shape of the retention characteristics ascertained by the form parameters $\alpha$ and $n_g$, and
- hydraulic conductivity and its anisotropy given by the saturated hydraulic conductivity $K_s$, the pore-connectivity l and the anisotropy $K_h^A$ (ration between horizontal and vertical hydraulic conductivity $K_h/K_v$).

Of these parameters only the water contents $\theta_r$ and $\theta_s$ impact exclusively water storage processes. The other parameters influence also the temporal discharge characteristic.

The base flow from the landfill during dry periods is sensitive to the pore-connectivity l and the form parameter $n_g$ of the matrix domain. Additionally the saturated hydraulic conductivity $K_s$ of both domains and the pressure head scaling factor $\alpha_h$ (channel domain) affect to some extent the base discharge.

Shape and amplitude of leachate generation peaks caused by heavy rainfall are mainly dependent on the

- saturated hydraulic conductivity $K_s$ of both domains, respectively their ratio,
- pore-connectivity l of the matrix domain,
- anisotropy $K_h^A$ ($K_h/K_v$) of the hydraulic conductivity of the matrix domain, and
- pressure head scaling factor $\alpha_h$ of the channel domain.

Furthermore, the retention characteristics of the matrix domain given by the form parameter $n_g$ and $\alpha$ influences the amplitude of discharge peaks.

The sensitivity of the leachate hydrograph to the saturated hydraulic conductivity $K_s$ of the channel domain becomes minor after the parameter exceeds a "certain" threshold. The value of this threshold is affected by the hydraulic characteristics of the matrix domain. A similar effect is observable for the pressure head scaling factor $\alpha_h$.

In Table 5-20 the influence of single model parameters on water storage, on base flow during dry periods and on leachate discharge peaks is assessed, whereby the degree of influence is divided into three categories:

– = weak
+ = medium
++ = strong

*Table 5-20* ***Influence of model parameters on the water flow through MSW landfills***

| | Parameter | *Water storage* | *Base flow* | *Discharge peaks* |
|---|---|---|---|---|
| Matrix domain | $\theta_r$ | ++ | – | – |
| | $\theta_s$ | ++ | – | – |
| | $\alpha$ | + | – | + |
| | $n_g$ | ++ | ++ | + |
| | $K_s$ | + | + | + |
| | l | ++ | ++ | ++ |
| | $K^A_h$ ($K_h/K_v$) | ++ | – | ++ |
| Channel domain | $\theta_r$ | – | – | – |
| | $\theta_s$ | – | – | – |
| | $\alpha$ | – | – | – |
| | $n_g$ | – | – | – |
| | $K_s$ | + | + | ++ |
| | l | – | – | – |
| | $\alpha_h$ | + | + | ++ |

Summarizing the outcome of the sensitivity analysis, the results of the water flow model are mainly dependent on the hydraulic properties of the matrix domain. The characteristics of the channel domain primarily the parameter $K_s$ and $\alpha_h$ becomes decisive for water flow, if they fall below a "certain" threshold. Altogether nine parameters (seven for the matrix and two for the channel domain) influence the water flow, whereby depending on the considered "flow process" (water storage, base flow, discharge peaks) different parameters are crucial.

## 5.4. *Evaluation of flow regimes*

### 5.4.1. Principles

The main aim for developing a new approach for modeling water flow through landfills is the need for a better insight into the hydraulic behavior of MSW landfills. Since water plays the key role in the metabolism of landfills, information on the transport of water is of direct interest for engineering stabilization processes or when evaluating the decomposition stage and predicting the duration of aftercare. Due to the highly heterogeneous nature of a landfill the flow field is not uniform. Internal structures of the landfill facilitate rapid water flow in restricted channel and voids, whereas large portions of the landfill are hardly participating in water flow (e.g. Zeiss & Major, 1993). This phenomenon is incorporated into the presented landfill model by dividing the waste mass into a matrix domain with slow water movement and a channel domain with fast water flow.

Sensitivity analyses for the model indicated that the following parameters influence the heterogeneity of the flow regime: $\alpha$, $n_g$, $K_s$, l, $K^A{}_h$ of the matrix domain and $K_s$, $\alpha_h$ of the channel domain. The values of these parameters can provide a first clue for assessing the uniformity respectively non-uniformity of the water flow. For instance large differences between the saturated hydraulic conductivity $K_s$ of channel and matrix domain, or high anisotropy $K^A{}_h$ of the hydraulic conductivity of the matrix domain result in heightened non-uniform water routing through the landfill.

Since the uniformity of the flow regime is influenced by seven hydraulic parameters, it is impossible to determine an overall quantitative hydraulic homogeneity grade for the water flow in landfills using the calibrated parameter values. However, such a hydraulic homogeneity grade quantifying the portion of waste mass participating in water flow would enhance insights into the metabolism of sanitary landfills. Current leachate emissions could be better evaluated regarding the stabilization stage of the whole landfill, since these reflect only the decomposition status of the preferential flow paths and their surroundings.

The calibrated hydraulic parameters of water flow simulations allow at the best only a qualitative evaluation of the water distribution. To obtain quantitative information about the portion of waste mass participating in water flow, knowledge on the flow velocity throughout the landfill is necessary. The simplest way to determine this characteristic is to perform solute transport simulations. In particular the discharge of conservative substances that are already dissolved in the leachate need to be modeled. In order to incorporate the hydraulic flow

regime of the considered landfill, the solute transport considerations must be based on the calibrated water flow model. The results of these simulations approximately represent the emission behavior of easy soluble salts (e.g. chloride or sodium) from the landfill.
Fortunately, the software HYDRUS-2D, on which the presented hydraulic landfill model is based, enables to simulate the transport of dissolved substances that are carried by water flow. The program uses the classical convection-dispersion-equation (Lapidus & Amundson, 1952) to compute the transport of solutes.

$$\frac{\partial c}{\partial t} = D\frac{\partial^2 c}{\partial z^2} - v\frac{\partial c}{\partial z}$$ ***Equation 5-1***

| | |
|---|---|
| $c$ | *Solute concentration [g m⁻³]* |
| $t$ | *Time [s]* |
| $D$ | *Dispersion coefficient [m² s⁻¹]* |
| $v$ | *Pore water velocity [m s⁻¹]* |
| $z$ | *Coordinate [m]* |

In order to focus the transport processes to convective transport and thus water flow only, hydrodynamic dispersion (caused by different length of flow paths and different flow velocities within the pores) was neglected during the solute discharge simulations. However, some dispersion had to be accepted in order to avoid numerical instabilities of HYDRUS-2D (Šimunek & van Genuchten, 2002).
Theoretical solute transport considerations (solute discharge from a homogeneous porous media - Figure 5-34) regarding and disregarding hydrodynamic dispersion are show in Figure 5-35 (piston flow versus hydrodynamic dispersion). Additionally the effect of heterogeneous flow conditions on the discharge of dissolved salts is presented in Figure 5-35.

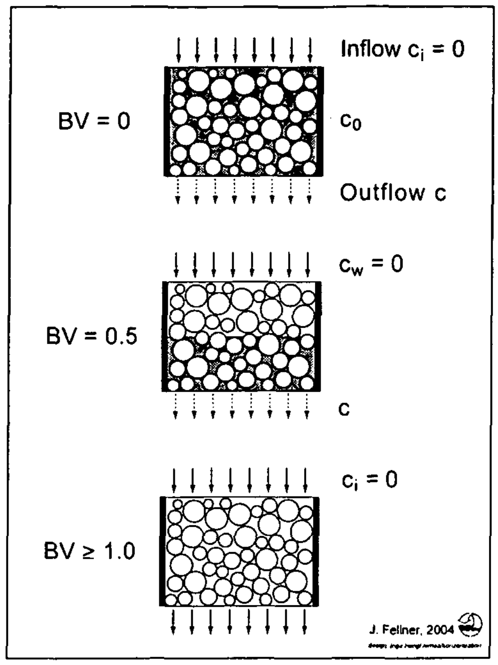



*Figure 5-34 Solute discharge from porous medium – sequence*

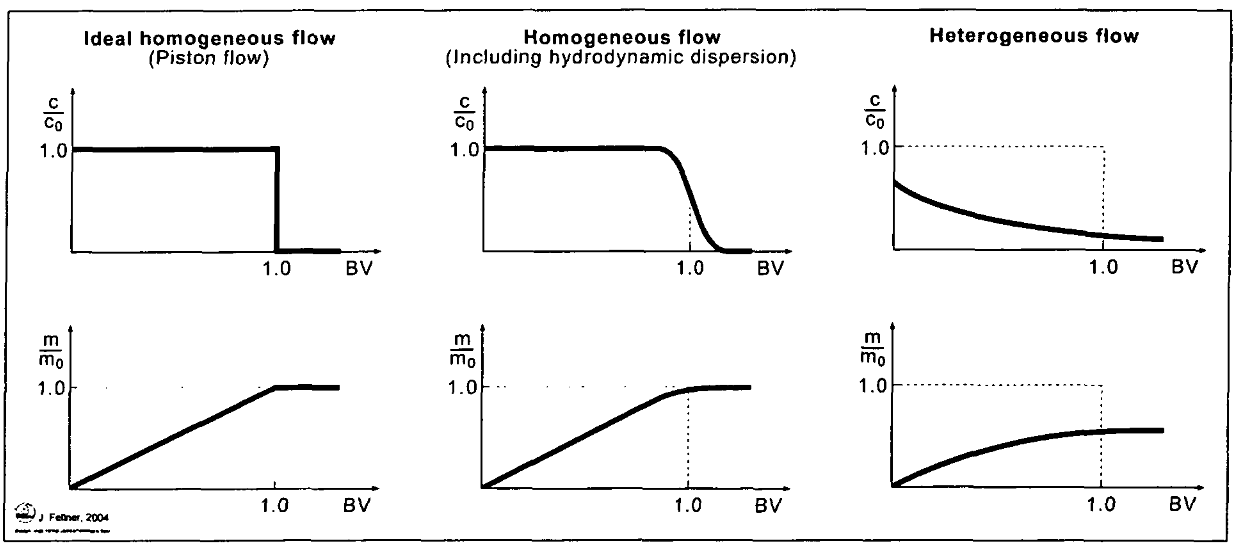



*Figure 5-35 Solute discharge (concentration and cumulative discharge) from porous media (piston flow, homogeneous and heterogeneous flow)*

$$BV = V_{tot} \cdot \theta$$ ***Equation 5-2***

| | |
|---|---|
| $BV$ | *Bed volume* |
| $V_{tot}$ | *Total volume of the porous media* |
| $\theta$ | *Volumetric water content* |

| | |
|---|---|
| $c_o$ | *Initial solute concentration of the pore water* |
| $c$ | *Effluent concentration* |
| $c_i$ | *Inflow concentration (=0)* |
| $m_o$ | *Initial solute mass of the pore water* |
| $m$ | *Discharged solute mass* |

The assumption of ideal homogeneous flow (piston flow) leads to a complete discharge of the initial solute mass (dissolved in the pore water) after a water exchange of one bed volume (BV). One bed volume equals the total amount of water present in the porous media (see Equation 5-2). The effluent concentration c drops sharply to zero at this point.
Solute transport simulations regarding hydrodynamic dispersion show a smooth drop in concentration after one BV. Nevertheless, almost the total solute mass is discharged after a water exchange of one bed volume.
For heterogeneous flow conditions the effluent concentration drops already at the beginning of water input. After an exchange rate of one bed volume part of the initial solute load is discharged only.

Solute transport simulations with HYDRUS-2D were performed to determine the fraction of waste mass taking part in water flow (convective transport is significant higher than diffusive transport). It was assumed that the considered substance (any salt) is dissolved and uniformly distributed throughout the landfill. The simulations focused on water flow and its effect on the solute discharge. Thus, dissolution, adsorption and diffusion processes as well as the presence of immobile water have been disregarded. In order to incorporate a possible impact of the water application rate on the uniformity of water flow, the solute transport simulations were carried out using the average water input rate of the landfill considered.
The modeling resulted in the cumulative discharge of the substance (salt) versus time respectively applied water amount. Figure 5-36 gives the outcome for the investigated landfills, whereby both cumulative solute discharge and applied water amount are standardized to the initial solute mass and the water amount stored inside the landfill. This scaling leads to the presented graph of normalized cumulative solute discharge versus bed volumes. Since only convective transport of dissolved compounds has been considered, the normalized cumulative solute discharge corresponds to the fraction of waste mass participating in "convective" water flow.

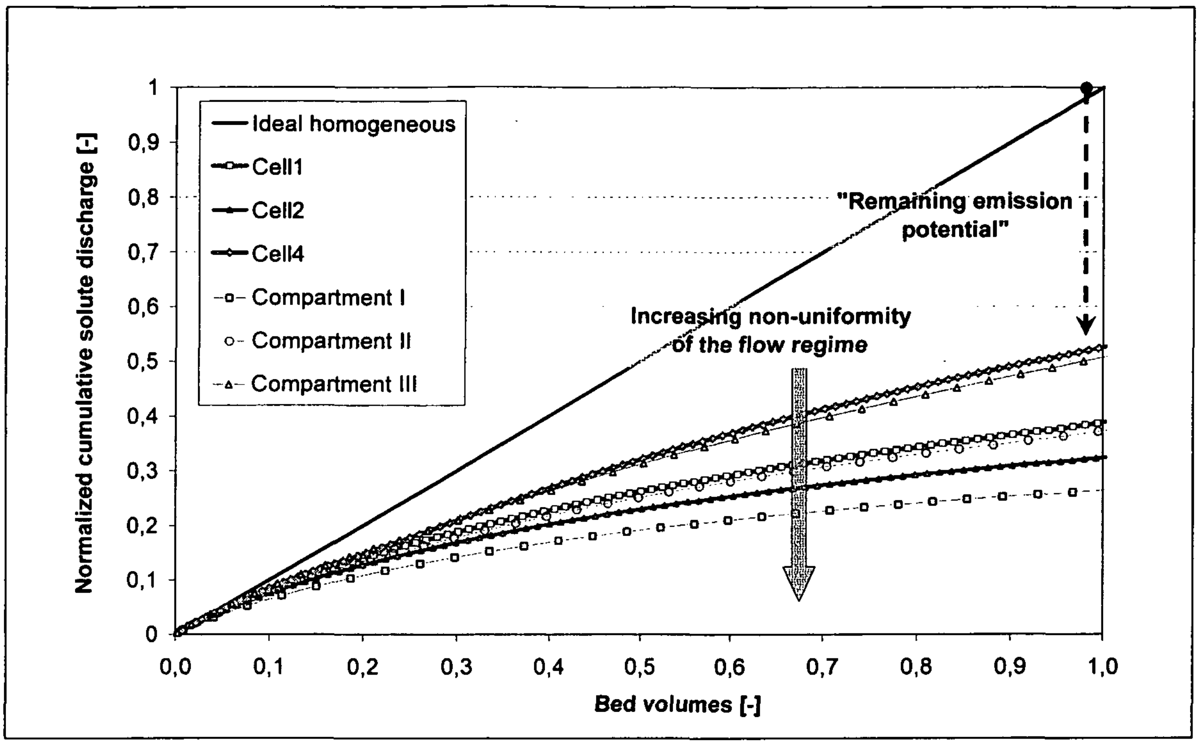



***Figure 5-36*** ***Cumulative solute discharge (normalized) versus leachate flow (expressed in bed volumes) - comparison of all simulated landfills***

Figure 5-36 indicates that after one bed volume of water percolated through Compartment I of the Breitenau landfill only 26 % of the initial solute mass has been discharged. Under ideal homogeneous flow conditions (piston flow) the total amount of initial solute mass would have been flushed out after an exchange rate of one bed volume. The figure provides quantitative information on the heterogeneity of water flow. For Compartment I (Breitenau landfill) it can be stated that less than 30 % of the total landfill mass is participating in water flow (extrapolated graph), which means that even after high water exchange rates (corresponding to a long time period) more than 70 % of the initial pollution load (salt) is still remaining inside the landfill. Thus, observed leachate quality at Compartment I reflects only the decomposition stage of less than 30 % of the landfilled waste. Changes in the flow paths could lead to sudden increase in leachate concentration. That implicates that low concentration values at Compartment I which may be already "compatible" with the environment, cannot be used as indicators for the end of the aftercare period. The remaining emission potential must be taken into account to evaluate the stabilization status and thus, the end of aftercare measures. In general it can be stated that the aftercare period (time starting from landfill closure till the potential pollution load is removed) is extended by highly non-uniform water flow as prevailing at Compartment I. Additionally to the amount of water that passed the landfill, temporal changes of the water paths become significant for the stabilization of the whole waste mass.

### 5.4.2. Comparison of modeling results

A comparison of the results of all simulated landfills (Figure 5-36) demonstrate that water flow in Compartment I is most non-uniform, followed by test Cell 2 of the Spillepeng site and Compartment II of the Breitenau landfill. Water flow in Cell 1 is comparable to those in Compartment II. Most homogeneous water routing is observable at Cell 4 and Compartment III. There the portion of waste mass participating in water flow amounts more than 50 % after a water exchange of one bed volume.
Considering the discharge rates given in Figure 5-36 it must be kept in mind that easy soluble salts have been investigated. For substances undergoing biochemical decomposition (e.g. nitrogen) and/or adsorption it can be assumed that discharge rates are significant lower. For these substances different parameters can limit their discharge, since the degradation processes depend on various factors (water exchange, water content, pH, redox-potential, composite of nutrients, ...).
The results show that the information content of leachate quality regarding the stabilization stage of the whole landfill is best for Compartment III and Cell 4 compared to the other landfills that have been investigated. For these two landfills the collected leachate is representative for the decomposition stage of at least 50 % of the landfilled waste mass after a water exchange rate of one bed volume.

Significant differences in the non-uniformity of the water flow (Figure 5-36), as denoted for all three compartments of the Breitenau landfill, as well as for Cell 2 and Cell 4 of the Spillepeng test cells, are confirmed by investigations concerning leachate quality.

#### 5.4.2.1. Comparison of leachate quality (Breitenau landfill)

Figure 5-37 shows the development of the Sodium concentration versus cumulative discharge at the three compartments of the Breitenau landfill. The decrease of concentration values is highest for Compartment I, whereas Compartment III shows a relatively high concentration level of Sodium (1000 mg/l) after a total leachate discharge (water exchange) of

105 l/ton MSW. Identical cumulative water exchange at Compartment I results in Sodium concentrations of around 600 mg/l. Due to the fact that similar waste was landfilled, these significant differences in leachate concentration inevitably ascribe to diverse water routing. Slow decrease in leachate concentration as observed at Compartment III indicates a more uniform water flow, as a larger waste mass contributes to leachate pollution. Whereas rapid decrease of soluble compounds in the leachate (Compartment I) refers to preferential flow paths, that short a large bulk of waste. Low leachate concentration values in this case represent the favored flow paths and their surroundings only.

The heterogeneity of Compartment II lies according to its leachate characteristics in between the two other compartments. These results confirm the findings of the mathematical modeling (see Figure 5-36).

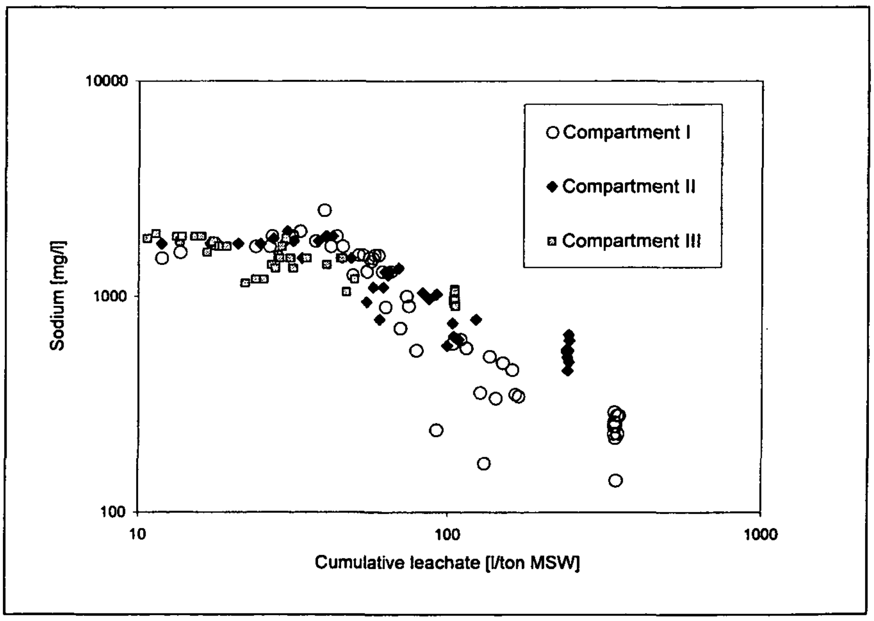



*Figure 5-37* ***Leachate quality (Sodium) versus cumulative leachate discharge (Breitenau landfill)***

### 5.4.2.2. Comparison of leachate quality (Spillepeng test cells)

The comparison of the flow regimes for Cell 2 and Cell 4 (Spillepeng test cells) had to be conducted differently due to insufficient records of the leachate concentration over the disposal time. Reliable information however, on the leachate quality over short periods (two years) was available. These data were used to investigate the influence of the discharge rate on the leachate concentration of easy soluble salts, since their emission behavior is only dependent on the water flow. The results of these investigations (Figure 5-38) denote that Chloride concentrations at Cell 2 are strongly influenced by the actual discharge rate. The dependency of the leachate quality at Cell 4 is significantly lower. For instance an increase of the leachate generation rate of one magnitude (0.1 to 1.0 mm/d) results for Cell 2 in a concentration decline of easy soluble compounds of around 50 %. For Cell 4 a reduction of only 20 % is observable. This fact can be seen as clear indication for diverse water routing in both cells. Uniform water movement in landfills is associated with slight influence of discharge rates on the leachate concentration, while the generation rate of leachate has major impact on its concentration in a flow regime of heightened heterogeneity. Consequently water flow in Cell 2 is more non-uniform compared to Cell 4. A result, that coincides with the outcome of the water flow modeling and thereon based solute discharge simulations.

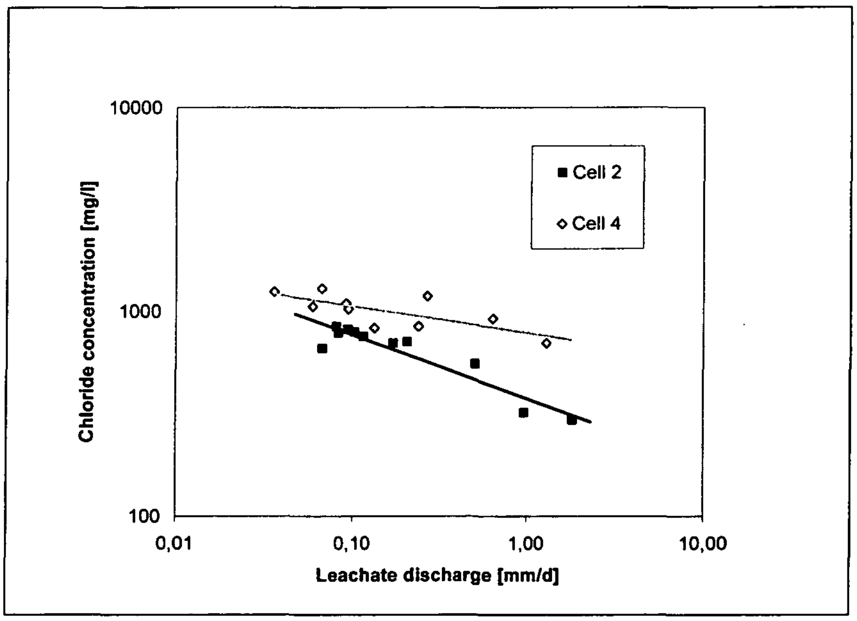



***Figure 5-38 Leachate quality (Chloride) versus leachate quantity (Cell 2 and Cell 4)***

The presented examples demonstrate the capability of the introduced landfill model for evaluating the homogeneity of the water flow inside a landfill. Quantitative information on the flow regime is essential for estimating the decomposition stage of the landfill using observed leachate quality. Also future emission behavior can only be predicted reliably, knowing the portion of the waste mass participating in water flow.

# 6. Summary and Conclusions

## *6.1. Introduction*

The main aim for developing a model for simulating water flow in MSW landfills was the need for a better insight into the black box "landfill". Since water plays the key role in the metabolism of landfills (e.g. Poland, 1975; Straub & Lynch, 1982b; Ehrig, 1983; Aragno; 1989) the mapping of water transport is of direct interest for engineering stabilization processes or when developing models for predicting leachate quality and biogas production.

## *6.2. Summary*

Up to now the prevailing approach for modeling water flow processes in solid waste media relied on the assumption of a homogeneous porous media (e. g. Straub & Lynch, 1982a; Schroeder et al., 1984; Korfiatis et al., 1984; Dematracopouls et al., 1986; Vincent et al. 1991; Noble & Arnold, 1991; Al-Yousfi et al., 1992; Demirekler et al., 1999). However, due to the heterogeneous nature of the waste media itself, the horizontal texture of the landfill caused by the landfilling and compaction technique, and construction elements with different hydraulic characteristics (such as gas wells or daily cover layers), the assumption of a homogeneous flow regime may be questioned. In several field investigations (e.g. Wiemer, 1982; Blight et al., 1992) it has been shown that the water content varies from saturated conditions to complete dryness inside the landfill. Preferential flow paths that short a large bulk of the landfill explain this. According to Ehrig (1983) and Uguccioni & Zeiss (1997) the rapid flow in those favored flow paths is believed to be the reason why existing landfill models are not in agreement with actual field observations. Also, the spatial and temporal variations in the leachate composition reported in the literature (El-Fadel, 1997; Åkesson & Nilson, 1997, Döberl et al., 2002) may partly attribute to non-uniform water flow.

In recently developed water flow models for MSW (Uguccioni & Zeiss, 1997; Bendz, 1998; Obermann, 1999) the heterogeneous character of landfills was taken into account. The waste body was not considered as a homogeneous media anymore. It was split into a channel domain with rapid water flow surrounded by a matrix domain with slow water movement. The mathematical approaches for describing the water flow in the two domains are different. Uguccioni & Zeiss (1997) used the model PREFLO (Workman & Skaggs, 1990) to simulate the water movement. This model assumes that the rapid flow in the channel domain follows Poiseuille's Law (1841) and the lateral water transfer from the channels into the matrix occurs according to Richards' Law (1931). Bendz (1998) used another assumption for describing the fast water flow in channels. A power function (kinetic wave model), as it has already been proposed by Beven & German (1981) to describe the water flow in macroporic soils, was used to determine the channel flow in landfills. Water filtrates into the matrix domain under wet conditions and is released again during dry periods. Obermann (1999) suggested a two-domain approach with Darcy flux in both zones, in the matrix as well as in the channel domain. Fast channel flow occurs only if the water input exceeds the hydraulic conductivity of the matrix domain. The application of two-domain water flow models partly results in a better fit between predicted and observed leachate generation rates. However, a large number of unknown model parameters must be accepted using these approaches. Up to now simulations are limited to laboratory experiments only. A framework for scaling up and validating models at landfill size is lacking. This is mainly due to sophisticated mathematical formulations that complicate the incorporation of variable boundary conditions prevailing at landfill sites enormously. Furthermore, all introduced two-domain concepts have in common that they are derived from the framework that has been carried out for non-uniform water flow in soils. Compared to soils, landfills are more heterogeneous, which inevitably results in a bigger fraction of preferential flow. This fact was taken into account in existing model approaches by adjusting the decisive parameters. However, a comparison of the characteristics of non-uniform water movement in cracked soils and MSW landfills point out significant differences. Whereas in soils preferential flow becomes minor with increasing depth (Bundt et al., 2000) the opposite effect is observable in landfills, represented by accelerated transport of solutes (Rosqvist et al., 1997) and bigger variation in water content towards the landfill bottom (e.g. Yuen et al., 2001). Moreover, rapid flow in fissures and macro-pores of soils is limited to wetting periods, while landfills show significant channel flow even during dry periods.

The disregard of these basic differences in the water flow pattern of soils and MSW landfills led to the development of a new two-dimensional two-domain approach. The proposed water flow pattern (Figure 6-1), derived from findings of different landfill studies, has been realized using the software tool HYDRUS-2D (Šimunek et al., 1996). This model enables to simulate water flow, solute and heat transport in variably saturated porous media.

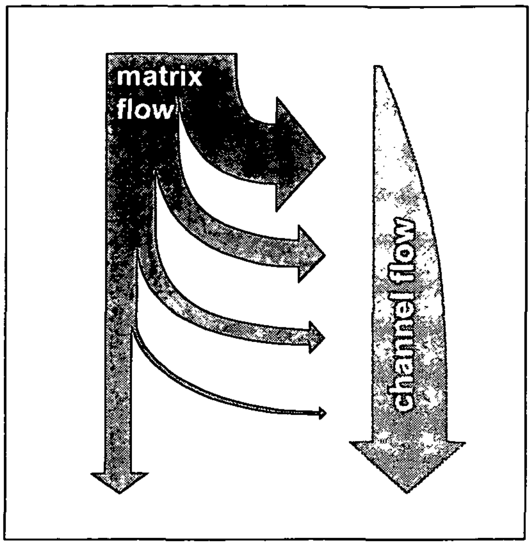



*Figure 6-1 General flow pattern of water in MSW landfills*

A two-dimensional flow field consisting of one vertical favored flow path surrounded by the waste mass (matrix domain) was defined using HYDRUS-2D. The preferential flow path (channel domain) was assigned a high permeability and a low or even no-retention capacity, while the matrix domain is characterized by low permeability and high retention capacity. Water flow in both domains is calculated according to Richards equation, whereby "virtual" suction power within the preferential flow path is assumed to ensure the proposed flow pattern. HYDRUS-2D accounts for variable boundary conditions (e.g. precipitation and evapotranspiration), which facilitates its application to field data. Runoff processes are not incorporated and must be considered separately.

Simulation results of the developed model concept were presented for two landfill sites: the experimental landfill Breitenau in Austria (~ 95,000 tons of waste), and the Spillepeng landfill in Malmö, Sweden (~ 25,000 tons of waste). Altogether leachate generation from six different waste compartments was simulated. The model results showed a good match with observed leachate generation rates. However, it was necessary to calibrate numerous parameters (14). Initial estimates of the parameters to calibrate were derived from hydraulic investigations at MSW landfills reported in the literature. By means of sensitivity analysis five parameters crucial for the overall storage of water inside the landfill were identified. These variables (residual and saturated water content: $\theta_r$ and $\theta_s$, form parameter $n_g$ of the retention characteristics, pore-connectivity $l$ and anisotropy $K^A_h$ of the hydraulic conductivity) characterize the hydraulic properties of the matrix domain only. Moreover, it was shown that also the non-uniformity of the water flow is mainly dependent on the characteristics of the matrix domain (anisotropy $K^A{}_h$, pore-connectivity l, form parameter $n_g$). Additionally to these parameters, the difference of the saturated hydraulic conductivities $K_s$ between both domains as well as the scaling factor for the pressure head $\alpha_h$ ("virtual suction power") have major impact on the shape of discharge hydrographs, and thus, the heterogeneity of the water flow in landfills.

Quantitative information on the uniformity of the water flow in landfills was obtained by solute transport simulations that are based on the calibrated water flow model. Thereto the discharge of a conservative substance was computed using HYDRUS-2D. In order to focus on water flow only, the solute discharge modeling was restricted to convective transport. The simulations result in graphs that provide the fraction of the initial solute load that has been discharged over time. This corresponds to the portion of waste mass participating in water flow versus cumulative water exchange rate (see Figure 6-2).

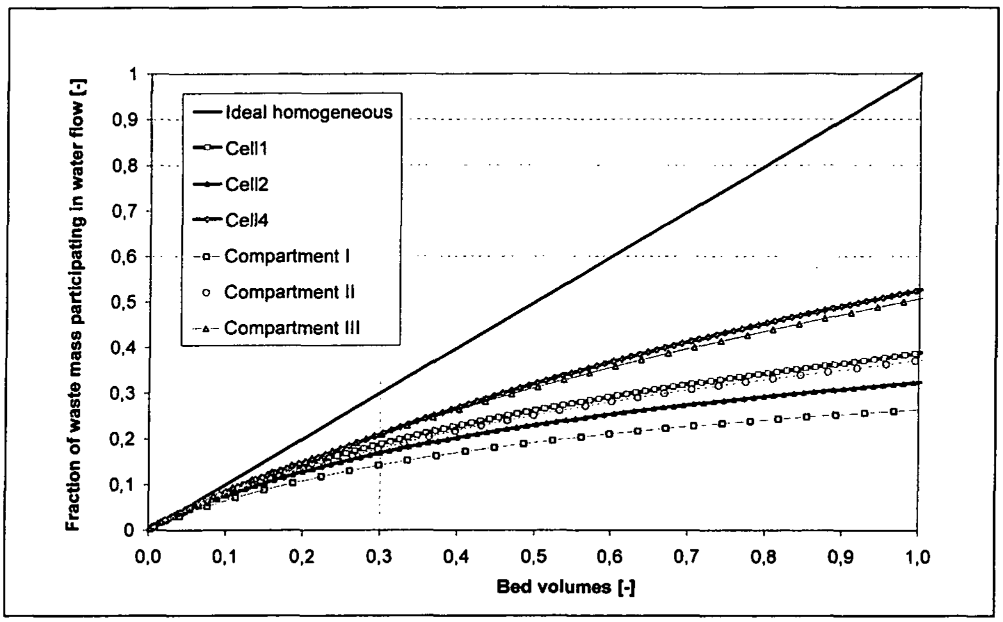



***Figure 6-2*** ***Fraction of waste mass participating in water flow versus water exchange (expressed in bed volumes)***

The results of the solute discharge simulations (Figure 6-2) indicate that the water flow in Compartment I (Breitenau landfill) is highly non-uniform, whereas Cell 4 shows the most uniform water flow of the investigated landfills. However, at most 50 % of the waste mass is participating in water flow after an exchange rate of one bed volume. Although, similar waste was landfilled and compacted in the same way (for each case: Breitenau landfill and Spillepeng test cells), significant differences in the uniformity of the water transport have been identified. Different water flow in the considered landfills results obviously in unlike metabolisms. Degradation processes of the waste mass excluded from water exchange are strongly decelerated (e.g. Klink & Ham, 1982; Bogner & Spokas, 1993), as water is the only carrier of substances within a landfill and only water flow facilitates the redistribution of chemicals, micro-organisms and nutrients.

The new model concept enables to quantify the hydraulic homogeneity of landfills, which leads to a better understanding of the metabolism and future emission behaviour of sanitary landfills.

When applying the software HYDRUS-2D for simulating water flow and solute transport, it was noticed that the model shows numerical problems for highly non-uniform flow regimes. Nevertheless, numerical instabilities could be avoided defining small meshed grids (spacing of less than 10 cm), in particular at the interface between matrix and channel domain.

## *6.3. Conclusions*

The differences in water flow pattern of heterogeneous soils and MSW landfills are not only important for mathematical modeling, but also for landfill engineering. The operation of modern landfills as "flushing bioreactor" is recommended by several researchers (e.g. Gronow, 1993; Reinhart & Townsend, 1998; Beaven & Knox, 1999). The purpose is to achieve a stable landfill within one generation (30 years). Enhancement of biochemical degradation processes as well as flushing of easily soluble compounds is the main objective of this strategy. However, preferential pathways that short a large bulk of waste mass must be accounted for when applying this method. It is shown in seceral investigations that water flow becomes more non-uniform towards the landfill bottom (e.g. Rosqvist et al., 1997). Thus, an evenly two-dimensional water application directly underneath the landfill cover, as promoted by Drees (2000), results in less waste exposed to the flushing water. In order to increase the participating water volume it is suggested to inject water in different depths, whereby an augmented number of feeding points is needed in bigger depths. Despite better insights into the landfill reactor, and probably improved water feeding, the operation strategy of flushing bioreactor must be questioned due to the huge water consumption, and thus, the enormous costs for leachate treatment even if part of the leachate is recirculated.

Existing approaches for assessing the remaining pollution potential and the future emission behavior of MSW landfills use either the observed actual leachate quality or the amount of water passed through the landfill (e.g. Allgaier & Stegmann, 2003) as the main indicator for the decomposition stage of landfills. However, as a major part of water flowing in landfills is restricted to favored flow paths, thereby bypassing large bulk of waste, observed leachate quality reflects only the flow paths and their surroundings. Sudden changes in the physical structure of the landfill (e.g. settlement due to biodegradation) may change the water routes. Thus, new parts of the landfill may be exposed to moving water. Consequently, the quality of the leachate may increase. Also the stabilization indicator "water amount passed through the landfill" (liquid to solid ratio) neglects the effect of preferential flow, associated with zones of high water exchange, and zones of nearly no exchange with high remaining pollution potential. In order to improve prediction of leachate quality and assessment of remaining pollution potential the magnitude of the true volume participating in the water flow through a landfill must be evaluated. The introduced landfill model based on HYDRUS-2D enables to estimate this volume by solute transport simulations for easy soluble salts. The model results show that the fraction of waste mass taking part in water flow varies considerably even in landfills of similar waste. Observed leachate qualities at landfill sites can be evaluated better regarding the degree of stabilization taking the prevailing flow conditions into account. For instance, equal leachate concentration levels could denote highly different decomposition stages of landfills (e.g. Compartment I versus Cell 4), since different fractions of the total waste mass may be reflected by the leachate. Also, existing assessment tools for landfill stabilization that are based on cumulative leachate quantity can be advanced by incorporating information on the water flow. The results for landfills with highly non-uniform water flow (e.g. Compartment I, see ) indicate that temporal changes of the water paths become important for the duration of the aftercare.

In addition to a better assessment of the landfill stabilization the new model allows comparing the hydraulic homogeneity of different landfills. Also the uniformity of water flow in small-scale experiments (e.g. landfill simulation reactors) can be determined using this method. When comparing the fraction of waste mass participating in water flow in full size landfills and small waste columns, the capability of laboratory experiments for predicting leachate generation can be assessed.

Practical examples for the application of the new model will be provided in an ongoing research project "A New Method to Characterize the Stability of Old, Large Size Landfills" conducted in cooperation between the Institute of Water Quality and Waste Management, Vienna University of Technology and the Department of Waste Management, Technical University of Hamburg-Harburg.

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# 8. Appendix

## *8.1. Short Description of HYDRUS-2D*

(after Bonaparte et al., 2004)

HYDRUS-2D is a two-dimensional unsaturated flow model developed at the U.S. Salinity Laboratory (Šimůnek et al., 1999). The model also simulates heat flow and solute transport. The current model is an extension of the earlier unsaturated flow codes SWMS_2D and CHAIN_2D. At the time of this writing version 2.02 of HYDRUS-2D was the most current. The model may be purchased from the International Ground Water Modeling Center, Colorado School of Mines, Golden, Colorado or http://www.Mines.EDU/research/igwmc/software/igwmcsoft/. The documentation and a free demo version of HYDRUS-2D may be downloaded from http://www.ussl.ars.usda.gov/models/hydrus2d/htm.

HYDRUS-2D uses a finite element method to solve Richards' equation in a plane oriented either vertically or horizontally. The two-dimensional domain may take on any geometric shape. Because the model is two-dimensional, lateral flow and anisotropy may be simulated. A sink term is included in Richards' equation for removal of water via plant transpiration. Vapor flow cannot be simulated. The model has an option for allowing soil properties to be temperature dependent, and it also allows hysteresis and spatial variability through a scaling transformation (Vogel et al., 1991). The unsaturated hydraulic conductivity is calculated by either a Brooks-Corey, van Genuchten-Mualem, or modified van Genuchten method. Precipitation, runoff, ET, soil water storage, and percolation are included in the water balance. Precipitation and potential evaporation are the only climatic inputs required. HYDRUS-2D does not have an option for internally calculating potential evaporation, so the user must use another model or method to generate data to input. Vegetation parameters required include the heads between which transpiration occurs and also the heads between which transpiration is optimal. A menu containing a variety of properties for plants is available. The distribution of roots must also be specified. Input required for soil properties includes saturated hydraulic conductivity and fitting parameters from the selected soil-water retention function. A menu of soil properties is available. In addition, van Genuchten properties can be predicted by inputting the percentage of sand, silt and clay, density, field capacity, and/or wilting point water content. HYDRUS-2D also has the option for inverse estimation of soil hydraulic properties from measured flow data.

The two-dimensional profile is created through a pre-processing module called Meshgen2D within the HYDRUS-2D graphical user interface. After the domain geometry is defined, Meshgen2D assists in generating the finite element mesh.

Boundary conditions may be specified flux, specified pressure head, unit gradient, atmospheric, seepage face, or deep drainage. Precipitation and potential evaporation are specified using the atmospheric option, which allows the boundary condition at the soil surface to change from either prescribed flux or prescribed head. The user inputs the upper and lower limits of head for which the prescribed flux boundary operates. Therefore, evaporation and precipitation will proceed at the potential rate until the soil surface dries or wets to a specified head. Once below the specified head, the boundary changes to a prescribed head boundary condition, and evaporation is limited by the ability of water to flow to the surface. If the surface becomes saturated during precipitation, excess precipitation is removed as runoff. The seepage face option allows water to exit the domain when the soil adjacent to the boundary becomes saturated. Deep drainage provides an option for a variable flux depending on the level of the groundwater table. Initial conditions may be specified as either water contents or pressure heads.

The HYDRUS-2D post-processor allows a variety of options for viewing output. Results can be displayed graphically, including an animation of changes in pressure head or water content through time. Cross-sections plotting pressure head or water content vs. depth or length may be taken from the profile at any time of the simulation. Other output options include viewing the instantaneous or cumulative water boundary fluxes over time, run time information, graphical display of soil hydraulic properties, or converting output to ASCII format.

## 8.2. *Retention models*

van Genuchten model (1980):

$$\theta(h) = \theta_r + \frac{\theta_s - \theta_r}{\left[1 + (\alpha \cdot h)^{n_g}\right]^{1-1/n_g}}$$ **Equation 8-1**

$$S_e = \frac{\theta(h) - \theta_r}{\theta_s - \theta_r} = \left[1 + (\alpha \cdot h)^{n_g}\right]^{-1+1/n_g}$$ **Equation 8-2**

$$K(S_e) = K_s \cdot S_e^{\,l}\left\{1 - \left[S_e^{\,n_g/(n_g-1)}\right]^{1-1/n_g}\right\}^2$$ **Equation 8-3**

Modified van Genuchten model (Vogel & Cislerova, 1988)

$$\theta(h) = \begin{cases} \theta_a + \dfrac{\theta_m - \theta_a}{\left(1 + |\alpha \cdot h|^{n_g}\right)^m} & h < h_s \\ \theta_s & h \geq h_s \end{cases}$$ **Equation 8-4**

Brooks and Corey (1964)

$$S_e = \frac{\theta(h) - \theta_r}{\theta_s - \theta_r} = (\alpha \cdot h)^{-\lambda}$$ **Equation 8-5**

$$K(S_e) = K_s \cdot S_e^{\frac{2+3\lambda}{\lambda}}$$ **Equation 8-6**

Campell (1974) und Huston & Cass (1987)

For $\theta \leq \theta_c$: $$h = a\left(\frac{\theta}{\theta_s}\right)^{-b}$$ ***Equation 8-7***

For $\theta \geq \theta_c$: $$h = \frac{a\left(1-\frac{\theta}{\theta_s}\right)^{1/2}\left(\frac{\theta_c}{\theta_s}\right)^{-b}}{\left(1-\frac{\theta_c}{\theta_s}\right)^{1/2}}$$ ***Equation 8-8***

whereby $$\theta_c = \frac{2b\theta_s}{1+2b}$$ ***Equation 8-9***

$$h_c = \frac{2b}{1+2b}$$ ***Equation 8-10***

$$K(\theta) = K_s\left(\frac{\theta}{\theta_s}\right)^{2b+2+p}$$ ***Equation 8-11***

| | |
|---|---|
| $\theta$ | *volumetric water content [$m^3$ $m^{-3}$]* |
| $\theta_s$ | *saturated volumetric water content [$m^3$ $m^{-3}$]* |
| $\theta_r$ | *residual volumetric water content [$m^3$ $m^{-3}$]* |
| $h$ | *suction (pressure) head [m]* |
| $\alpha$ | *form parameter [1/m], inverse bubbling pressure* |
| $n_g$ | *form parameter [-]* |
| $K$ | *hydraulic conductivity [m $s^{-1}$]* |
| $K_s$ | *saturated hydraulic conductivity [m $s^{-1}$]* |
| $l$ | *pore-connectivity parameter [-]* |
| $S_e$ | *degree of saturation [-]* |
| $\theta_a$ | *$\leq \theta_r$ extrapolated volumetric water content [$m^3$ $m^{-3}$] at infinite small matrix potential* |

$\theta_m$ — $\geq \theta_s$ *extrapolated volumetric water content* [$m^3$ $m^{-3}$], *at full water saturation*

$\theta_r$ — *volumetric water content* [$m^3$ $m^{-3}$] *at the hydraulic conductivity* $K_k$

$\lambda$ — *empirical pore coefficient [-]*

$K_k$ — *hydraulic conductivity* [$m$ $s^{-1}$] *at a water content of* $\theta r$

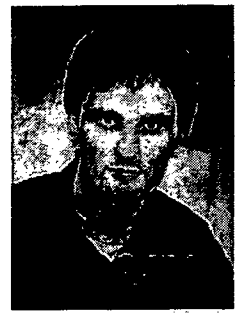
$L$ — *empirical tortuosity coefficient according to Mualem (usually 0.5)*

$a$ — *air entry value [m]*

$b$ — *Campell exponent [-]*

$p$ — *pore coefficient [-]*

# CURRICULUM VITAE



| | |
|---|---|
| Name: | Johann Fellner |
| Date of birth: | 06.03.1974 |
| Place of birth: | Neunkirchen, Austria |

**Education:**

| | |
|---|---|
| *2001 – 2004* | Doctoral student, Vienna University of Technology, Faculty of Civil Engineering, Institute for Water Quality and Waste Management |
| *1993 – 1999* | Master student, Land and Water Management and Engineering, Vienna University of Natural Resources and Applied Life Sciences. |
| | Master thesis: "Comparison of Solute Transport Models", Institute for Hydraulics and Rural Water Management |
| *1988 – 1993* | College for Engineering Wr. Neustadt, Electrical Engineering |
| *1980 – 1988* | Primary School St. Lorenzen, and Comprehensive Secondary School Neunkirchen |

**Work experience:**

| | |
|---|---|
| *Since 07/2003* | Senior research associate, Vienna University of Technology, Institute for Water Quality and Waste Management |
| *01 – 06/2003* | Visiting Researcher, Department of Water Resources Engineering, Lund University, Sweden |
| *10/2000 – 12/2003* | Research assistant, Vienna University of Technology, Institute for Water Quality and Waste Management |
| *10/1999 – 09/2000* | Civil service at the Neunkirchen hospital |
| *1998 – 1999* | Engineering work in the field of water and waste management, ÖKOTEC, Wr. Neustadt |