AVF-00/4

Projektrapporter

Solid waste and the water environment in the New European Union perspective
Process analysis related to storage and final disposal
Thesis

M Marques

Rapporterna kan beställas från
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Statens energimyndighet
Box 310, 631 04 Eskilstuna

Energimyndigheten

00-11-22

Energimyndigheten

Titel:	Solid Waste and the Water Environment in the New European Union Perspective Process Analysis Related to Storage and Final Disposal	
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RAPPORT INOM OMRÅDET AVFALL/BIOGAS

Rapportnummer:	AVF 00/4
Projektledare:	William Hogland
Projektnummer:	P3904
Projekthandläggare på Statens Energimyndighet:	Bengt Blad

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SOLID WASTE AND THE WATER ENVIRONMENT IN THE NEW EUROPEAN UNION PERSPECTIVE

Process Analysis Related to Storage and Final Disposal

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Stockholm 2000

TRITA-KET R123
ISSN 1104-3466
ISRN KTH/KET/R--123--SE

KTH

KUNGL
TEKNISKA
HÖGSKOLAN

TRITA-KET R123
ISSN 1104-3466
ISRN KTH/KET/R--123--SE

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ACADEMIC THESIS

Under the auspices of the Royal Institute of Technology (KTH) in Stockholm this thesis is submitted for formal review for the degree of Doctor of Philosophy (PhD) in Chemical Engineering, Tuesday the 6th of June, 2000 at 2:30 p.m., Room D3, Lindstedtsvägen 5, KTH, Stockholm.
The Royal Institute of Technology appointed opponent is Professor Raffaello Cossu, Faculty of Engineering, Padova University, Italy. The thesis is to be publicly defended in English.

AKADEMISK AVHANDLING

Som med tillstånd av Kungliga Tekniska Högskolan i Stockholm framlägges till offentlig granskning for avläggande av teknologie doktorsexamen tisdagen den 6 juni 2000, kl 14:30 i Sal D3, Lindstedtsvägen 5, KTH, Stockholm. Fakultetsopponent är Prof. Raffaello Cossu, Faculty of Engineering, Padova University, Italy. Avhandlingen försvaras på engelska.

Title:	"Solid Waste and the Water Environment in the New European Union Perspective: Process Analysis Related to Storage and Final Disposal"
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Language:	English

ABSTRACT

Processes that occur during storage and final disposal of solid waste were studied, with emphasis on physical and chemical aspects and their effects on the water environment, within the New European Union perspective for landfilling (Council Directive 1999/31/EC of 26 April 1999). In the new scenario, landfilling is largely restricted; waste treatments such as incineration, composting, recycling, storage and transportation of materials are intensified. Landfill sites are seen as industrial facilities rather than merely final disposal sites. Four main issues were investigated within this new scenario, in field- and full-scale, mostly at Spillepeng site, southern Sweden.

Adequacy of storage piles. Regarding the increasing demand for waste storage as fuel, the adequacy of storage in piles was investigated by monitoring industrial waste (IND) fuel compacted piles. Intense biodegradation activity, which raised the temperature into the optimum range for chemical oxidation reactions, was noticed during the first weeks. After about six months of storage, self-ignition occurred in one IND pile and one refuse derived fuel (RDF) pile. Heat, O_2 and CO_2 distribution at different depths of the monitored IND pile suggested that natural convection plays an important role in the degradation process by supplying oxygen and releasing heat. Storage techniques that achieve a higher degree of compaction, such as baling, are preferable to storage in piles.

Discharge from landfill for special waste. Regarding changes in the composition of the waste sent to landfills and the consequences for its hydrological performance in active and capped landfills, discharge from a full-scale landfill for special/hazardous waste (predominantly fly ash from municipal solid waste (MSW) incineration) was modelled using the U.S. EPA HELP model. Hydraulic properties of the special waste were compared with those from MSW. Lower practical field capacity and higher hydraulic conductivity at special waste cells led to faster production of greater amounts of discharge, in response to infiltration (already during operation), than that observed in MSW cells. This feature must be considered when designing *on-site* leachate treatment systems for special/hazardous waste landfills.

Stormwater runoff and pollutant transport. The intensification of waste handling practices exposed to rainfall at waste management parks in the new scenario led to an investigation of about 22 constituents of stormwater runoff and pollutant transport from different areas/activities and roads within the Spillepeng site. The concentration values for some parameters in some areas and roads exceeded the concentrations found in leachate from covered landfill. Concerning chemical oxygen demand and nutrients, the stormwater from Spillepeng showed a higher range of median concentration values in the stormwater than is typical of ranges for roadways, and residential and industrial areas in Sweden. The reverse occurred for heavy metals, excluding copper.

Groundwater monitoring programmes. In particular, the adequacy of groundwater monitoring programmes at landfill sites, was investigated. Significant differences between up- and down-gradient wells and trends not visualized by direct inspection of time series data were detected with statistical analyses. The non-parametric rank-sum test was more powerful and robust than the *t*-test in detecting differences between up- and down-gradient paired monitoring wells. The seasonal Kendall test was more powerful than the Mann-Kendall test to detect trends for individual constituents. Non-parametric slope estimators and the Winter's Method were used to forecast the time needed to reach the EU mandatory limits for nitrate and ammonia in potable water. However, indications of aquifer heterogeneity suggest that these trends may reflect local effects, rather than a real improvement in, or degradation of, the groundwater quality. Nevertheless, the inclusion of statistical procedures in landfill monitoring programmes is suggested, as an additional useful tool.

Keywords: convective flow, groundwater pollution, leachate modelling, non-parametric statistics, self-ignition, solid waste, storage of fuel material, stormwater runoff, time series.

SOLID WASTE AND THE WATER ENVIRONMENT IN THE NEW EUROPEAN UNION PERSPECTIVE

Process Analysis Related to Storage and Final Disposal

Marcia Marques

PhD Thesis

VETENSKAP OCH KONST

KTH

Department of Chemical Engineering and Technology
Royal Institute of Technology

Stockholm 2000

To my mother Nena (in memoriam)

ABSTRACT

Processes that occur during storage and final disposal of solid waste were studied, with emphasis on physical and chemical aspects and their effects on the water environment, within the New European Union perspective for landfilling (Council Directive 1999/31/EC, 1999).

In the new scenario, landfilling is largely restricted; waste treatments such as incineration, composting, recycling, storage and transportation of materials are intensified. Landfill sites are seen as industrial facilities rather than merely final disposal sites. Four main issues were investigated within this new scenario, in field- and full-scale, mostly at Spillepeng waste management park, southern Sweden.

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Keywords: convective flow, groundwater pollution, leachate modelling, non-parametric statistics, self-ignition, solid waste, storage of fuel material, stormwater runoff, time series.

SAMMANFATTNING
(ABSTRACT IN SWEDISH)

De processer som uppträder i samband med lagring och slutlig deponering av kommunalt avfall studerades med tonvikt på fysiska och kemiska aspekter, och deras betydelse för vattenmiljön samt mot bakgrund av det nya EU-direktivet för deponering (Council Directive 1999/31/EC, 26 april 1999).

Med det nya Direktivet avser EU att kontrollera emissionerna från deponierna och förkorta avslutningsfasen. I det nya avfallsscenariot är deponering kraftigt begränsat och avfallshantering såsom förbränning, kompostering, återvinning, lagring och transport av material är intensifierad. Deponier kan idag hellre betraktas som industriella anläggningar än endast som platser för slutlig förvaring av avfall. Fyra huvudsakliga frågeställningar undersöktes i full skala i fält och mestadels på Spillepengs "Avfallshanteringspark" i södra Sverige, inom ramen för ovan nämnda scenario:

Lämpligheten av lagring i högar. Mot bakrund av ett ökat behov av lagring av avfallsbränslen undersöktes lämpligheten av lagring av industriavfall i kompakterade högar. Betydande biologisk aktivitet vilket kunde noteras under de första veckorna, ökade temperaturen till optimal nivå för kemisk oxidation. Efter ungefär sex månaders lagring inträffade självantändning i en av bränslehögarna med industri-avfall och i en hög med RDF (refuse derived fuel). Temperaturen, syre- och koldioxidfördelningen som mättes på olika djup i bränslehögen med industriavfall antyder att naturlig konvektion spelar en viktig roll i nedbrytningsprocessen genom att tillföra luft innehållande syre och avlägsna värme. Lagringstekniker där hög kompakteringsgrad uppnås såsom vid balning är att föredra framför lagring i högar.

Utsläpp av lakvatten från celler med specialavfall. Med avsikten att studera hydrologiska konsekvenser av det deponerade avfallets sammansättning i en aktiv respektive täckt och avslutad deponi modellerades avrinningen från fullskalecellen med specialavfall (huvudsakligen innehållande flygaska från förbränning av kommunalt avfall) med hjälp U.S. EPA HELP-modellen. Hydrauliska förhållanden för specialavfall jämfördes med de för kommunalt avfall. Lägre fältkapacitet och högre hydraulisk konduktivitet i cellen med specialavfall ledde till snabbare och större produktion av lakvatten som en respons på infiltrationen än vad som observerats i cellen med kommunalt avfall. Denna egenskap måste beaktas när man dimensionerar lokala behandlingssystem för lakvatten från deponier innehållande specialavfall.

Dagvattenavrinning och föroreningstransport. Intensifieringen av de avfalls-hanteringsmoment som exponeras för nederbörd i en "avfallshanteringspark" i det

nya scenariot föranledde en undersökning omfattande ca 22 föroreningsvariabler i dagvattnet och beräkning av föroreningstransporten från olika delytor/aktiviteter och vägar inom Spillepengsområdet. Föroreningskoncentrationen för några av parametrarna uppmätta inom en del av områdena och värdena uppmätta i dagvatten från vägarna översteg koncentrationen i lakvattnet uppsamlat från täckta celler. Vad gäller COD och näringsämnen så uppvisar dagvattnet på Spillepeng högre medianvärden på föroreningskoncentrationen än vad som är typiskt för vägområden, bostads- och industriområden i Sverige. Motsatsen uppmättes för alla analyserade tungmetaller bortsett från koppar.

Mätprogram för grundvatten. Mätprogrammets lämplighet för grundvattenstudier undersöktes speciellt. Signifikanta skillnader mellan uppströms och nedströms belägna grundvattenrör såväl som i trender som ej kunnat visualiserats genom direkta studier av tidsserier påvisades genom statistisk analys. Det icke-parametriska "rank-sum" testet var mer kraftfullt och resistent för extremvärden (outliers) än *t*-testet vad gäller detektering av differenser vid uppströms och nedströms parning av uppmätta grundvattennivåer. Vidare var det årstidsrelaterade Kendall-testet mer kraftfullt än Mann-Kendall-testet för att detektera trender hos enskilda föroreningsvariabler. Icke-parametriska gradientuppskattningar och Winters metod användes för prognostisering av den tid som behövs för att uppnå de av EU föreskrivna gränsvärdena för nitrat och ammoniak i dricksvatten. Emellertid gör indikationer på akvifärens heterogenitet gällande att trender kan reflektera lokala effekter snarare än förbättring eller försämring av grundvattnets kvalitet. Icke desto mindre föreslås användandet av statistiska metoder som ett extra hjälpmedel vid analys av mätserier för grundvatten.

Nyckelord: konvektivt flöde, grundvattenförorening, lakvattenmodellering, icke-parametrisk statistik, självantänding, fast avfall, lagring av bränslen, dagvatten-avrinning, tidsserier.

PREFACE

In a modern society that employs recycling and re-use based on the close-loop philosophy, a landfill site with a number of methods for waste treatment, storage and transportation of material is more like an industrial facility than merely a final disposal site of waste, and should be better called a "waste management park".

Following the new directive of the European Union regarding landfilling of waste decided on April 26, 1999 (Council Directive 1999/31/EC, 1999), landfilling of waste in Europe is now largely restricted. Simultaneously, recycling and recovery of materials and energy is encouraged according to the same directive to safeguard natural resources and obviate wasteful use of land. According to the directive, further consideration should be given to incineration of municipal and non-hazardous waste with energy recovery, composting and biomethanisation.

With the new Directive, the European Union intends to control landfill emissions and to shorten the after-care phase. Pre-treatment of degradable organic matter prior to disposal is, according to the Directive, an important procedure to control emissions.

According to the mentioned Directive, it is also necessary to establish common monitoring procedures during the operation and after-care phases of a landfill in order to identify any possible adverse environmental effect of the landfill and take the appropriate corrective measures. An improvement in the current monitoring procedures, regarding sampling and data analysis is expected.

One direct consequence of this new policy is that the amount of waste consigned to landfills will be reduced.

Another important consequence, easily foreseen, is that the composition of the waste consigned to landfills will change. Increasing production of combustion residues from incinerators must be used or landfilled, providing that the alternatives do not give rise to unacceptable environmental impacts or health hazards. Special/hazardous waste not suitable for recycling or re-use is likely to be the main waste stream destined for landfill.

A third relevant effect of the new waste management policy is that several waste handling options are likely to be intensified. Among these options it can be mentioned: sorting of slag for subsequent recycling of different fractions; composting of garden waste, polluted soil and sludge; wood chipping for wood-fuel production; and crushing and sorting of construction and demolition waste for road construction.

Storage of waste as fuel for incineration during the proper period of the year to meet the energy demands of different seasons is a procedure particularly important in cold climate regions. The expected increase in the amount of waste to be incinerated should lead to an improvement in the technology of the storage of waste without significant losses to its calorific value.

The present thesis deals with five issues: (1) geology and hydrogeology; (2) air, water and heat transport processes related to self-ignition of stored combustible materials; (3) water flow in unsaturated zones and discharge modelling; (4) urban hydrology and stormwater pollutant transport; and (5) statistical approaches for water quality time series analysis.

It was decided that the investigation should be carried out at field- or full-scale. The awareness of the fact that not all variables can be controlled in field- or full-scale was compensated for by the unique knowledge acquired during such investigation.

Due to the wide range of issues investigated and the selected scale, the "principle of parsimony" was applied; the simpler models, which reproduce or can explain the observed phenomena, were preferred over more complex models.

In this sense, the present investigation offers a more general overview than if it had only concentrated on one issue (e.g., water flow in unsaturated zones); however, it delves significantly deeper within each studied issue than it would have by using, for instance, a system analysis methodology.

The approach applied is intended to develop a scenario where selected important aspects related to waste treatment and disposal and the water environment are analysed within the new EU perspective, and at the same time, to improve the knowledge regarding specific aspects related to each selected issue.

The first chapter introduces the theoretical background handled during different parts of the project development. Particularly the sections related to the full-scale studies (1.2, 1.3, and 1.4) reflect the "principle of parsimony" mentioned above.

The second chapter describes the sites included in the present study: (a) the Spillepeng waste management park, particularly the landfill section, with emphasis on geology and hydrogeology aspects and, (b) the Lidköping incineration plant and its area of storage of baled waste included in the stormwater studies.

Finally, an overview of the five appended papers is followed by conclusions.

The following is a summary of the content of the papers.

- Physical, biological and chemical processes that occur during waste storage for later incineration of waste as fuel were investigated in **Paper I**. Two events of spontaneous combustion were utilized for investigation of such processes and theoretical background developed for storage of forest fuel available in the literature was applied.

- Concerning landfilling and water-related problems, **Paper II** addresses the hydraulic properties and the hydrological performance of special waste landfill cells (with mainly fly ash from MSW incineration), and compares these properties with those found in MSW and sludge co-disposal cells.

- The intensification of several waste handling activities in open areas in modern waste management parks will result in increasing pollutant loads transported by stormwater runoff. **Paper III** compares the water pollutant transported by stormwater runoff generated at several activities in Spillepeng with (a) the leachate composition from different landfill cells, and (b) the stormwater runoff from a number of industrial activities.

- In **Paper IV**, the stormwater runoff from different activities in a waste management park, including storage of baled waste is analysed within the Swedish context where stormwater runoff, typical for industrial, residential and traffic areas, are included.

- In **Paper V**, the adequacy of the current procedure of direct inspection of time series data for monitoring of groundwater quality at landfill sites is investigated. Non-parametric statistical procedures for time series data set analysis are applied to detect differences between up- and down-gradient monitoring wells, and to detect trends in the concentration of independent constituents.

APPENDED PAPERS

This thesis is based on the following papers, referred to by their Roman numerals from **I** to **V**:

I. Hogland, W. and Marques, M. 1999. Physical, biological and chemical processes during storage and spontaneous combustion of waste fuel. *Resources, Conservation and Recycling* (in press).

II. Marques, M. and Hogland, W. 1999. Hydrological performance of incineration residues co-disposed with other special wastes and MSW co-disposed with sludge in full-scale cells. *Waste Management* (in press).

III. Marques, M. and Hogland, W. 1999. Stormwater runoff and pollutant transport related to activities carried out in a modern waste management park. *Waste Management and Research* (in press).

IV. Marques, M. 2000. Stormwater runoff pollution at waste management sites compared to runoff from other land uses (submitted).

V. Marques, M. 2000. Statistical analysis of groundwater quality time-series in a landfill site (submitted).

OTHER PAPERS

Other papers not included in the thesis, but which were produced during the doctoral programme and are related to the subject of the thesis, are:

1. Hogland, W., Marques, M. and Thörneby, L. 1998. Landfill mining - space saving, material recovery and energy use. *In: Proceedings of the Seminar on Waste Management and the Environment-Cooperation Between Nordic Countries and Countries in the Baltic Sea Region,* 5-7 November 1997, Kalmar, Sweden. Kalmar University, Kalmar, Sweden, pp. 339-355.

2. Marques, M. and Hogland, W. 1999. Leachate modelling in full-scale cells containing predominantly MSW incineration residues. In: Christensen, T.H., Cossu, R., and Stegmann, R. (Eds), *Proceedings of The Seventh International Waste Management and Landfill Symposium – Sardinia '99*, 4-8 October 1999, Cagliari, Italy, Vol. I. CISA, Cagliary, Italy, pp. 613-620.

3. Marques, M. and Hogland, W. 2000. Surface runoff quality in regard to different activities carried out at a modern landfill in Sweden. In: *Proceedings of the ISWA World Congress Paris 2000,* 3-6 July 2000, Paris (in press).

4. Hogland, W., Marques, M., Nimmermark, S. and Larsson, L. 2000. Landfill mining and stage of waste degradation in two landfills in Sweden. In: Proceedings of the *Asian-Pacific Landfill Symposium - APLAS Fukuoka 2000,* 11-13 October 2000, Fukuoka, Japan (in press).

5. Marques, M. 2000. Leaching tests with aged fly-ash from MSW incineration compared to leachate time-series data in full-scale cells (working paper).

6. Marques, M. 2000. Environmental risk assessment of a landfill site with hydraulic trap design: Modelling of water flow and pollutant transport (working paper).

ACKNOWLEDGEMENTS

About three and a half years ago, I came to Sweden to attend a doctoral programme, at the invitation of Professor William Hogland. As my supervisor, he has supported me with guidance and encouragement during all stages of my research work. As a friend, he has helped me face the normal difficulties associated with living abroad. For both reasons, he will always have my loyalty and gratitude.

I am honoured to have Professor Ivars Neretnieks as examiner, from the Department of Chemical Engineering and Technology, Royal Institute of Technology–KTH.

My doctoral studies began at the Department of Water Resources Engineering, Lund University–LU, and were complemented and finalized at the Department of Chemical Engineering and Technology, Royal Institute of Technology–KTH. The writing activity was developed primarily at the Department of Technology, Kalmar University–HIK, following the appointment of Professor Hogland to the chair of that Department. I am truly thankful that these three institutions made me feel so welcome.

The Swedish National Energy Administration–Energimyndigheten, and formerly the Swedish National Board for Industrial and Technical Development–NUTEK, granted important financial support. The Swedish Institute made it possible for me to come to Sweden to pursue my studies in the first instance.

My employees in Brazil–the Department of Environmental Engineering, Rio de Janeiro State University, the research company Pesagro-Rio and the Guanabara Bay Clean-up Programme–provided me with the necessary licences to study in Sweden.

Most of the fieldwork was carried out at Spillepeng landfill in Malmö, Sweden, operated by SYSAV AB. I am deeply indebted to Mr Håkan Rylander, President of SYSAV AB, who opened the doors of the company to me. Many thanks are also directed to the personnel at Spillepeng, particularly Maria Björngreen and Ragnar Dyrlund-Kristiansen, for making the necessary data available to me. The cooperation of Lidköpings Värmeverk AB, responsible for the Lidköping district heating plant, is also acknowledged.

Professor Alexander Lauber (HIK) supplied me with optimism and encouragement during our interesting discussions about science and everyday topics.

During the fieldwork, in windy-cold Scandinavia, I received help from my colleague Lars Thörneby. Lars was also an excellent roommate, later on replaced by another outstanding and supportive companion, Viatcheslav Moutavtchi–"Slava".

In my struggle with computers, I was supported at different times by Stefan Petersson, Claes Johansson, Staffan Karlsson and Magnus Eriksson (HIK), and Lenart Grahan (LU).
The secretaries Annika, Christine, Camilla (HIK), Kerstin and Clair (LU) helped me with administrative matters, and the kindnesses that we all need.

Since Leonardo was born, there has not been a single day that I have not thought how lucky I am for having him as a son. With his enlightened mind and fighting spirit, he has been the main inspiration and driving force behind me being able to accept new challenges.
To Leonardo and my dearest family, Miriam, Daniel and Liane, thank you for sharing this life with me and for being there whenever I need you.

April 2000

CONTENTS

1. THEORETICAL BACKGROUND

1.1 Storage of Waste as Fuel

The mass of municipal solid waste (MSW) incinerated annually in EU countries amounts to about 26 million tonnes (OECD, 1997), and this value is likely to increase considerably over the coming years. Furthermore, local legislation prohibits the incineration of waste without energy utilization. In northern Europe, energy recovery from waste is normally used to produce hot water or steam for heating, demand for which is unevenly distributed over the year. Seasonal storage of waste fuel is therefore necessary. In the rest of Europe, the energy recovered is usually converted into electricity. The level of construction of new incineration plants, as well as increases in the capacity of existing plants, is not trivial. Apart from the considerable investment involved, environmental legislation and regulations regarding land use place constraints on the development of such plants. In addition, other problems have been identified from trials aimed at increasing the amount of waste that can be recycled. Thus, improved knowledge concerning physical, chemical and biological processes occurring during waste storage is urgently required.

Hundreds of landfill fires are reported annually in Sweden (Naturvårdsverket, 1994), and in Finland (Ettala et al., 1996). These waste fires lead to emissions of carcinogenic and mutagenic substances, such as polyaromatic hydrocarbons (PAHs), chlorinated monoaromatic compounds, polychlorinated biphenyls (PCBs), and polychlorinated dibenzo-dioxins and -furans (PCDDs and PCDFs) (Ruokojärvi et al., 1995a and 1995b).

From controlled experiments with MSW, it has been reported that five times more PCBs are released from fires than from incinerator plants equipped with flue gas cleaning equipment (Bergström and Björner, 1992). The total amount of dioxins released in these experiments was about 1 mg per tonne of waste, which is four times the amount released from incinerators.

Storage of household waste under different conditions for one year showed no increase in temperature above 70 °C in piles of heights lower than 3 m (Bramryd et al., 1990; Åkesson et al., 1991). Nevertheless, there have also been reports of spontaneous combustion in piles of wood fuel (Chalk, 1968; Cole, 1972).

Field experiments, including tests of several methods of storage of different categories of waste have been carried out (Hogland et al., 1996; Hogland, 1998; Tamaddon et al., 1995; Hogland et al., 1993).

Studies concerning the storage of fuel have been previously carried out for forest material (Ernstson, 1995; Ernstson and Rasmuson, 1993; Ernstson et al., 1991;

Ernstson and Rasmuson, 1990; Thörnqvist, 1988; Kubler, 1987; Collin et al., 1986, Colling et al., 1985) from where most of the theory presented here was derived.

1.1.1 Physical, biological and chemical and processes

Spontaneous combustion is an exothermic process that requires combustible material, an elevated temperature and oxygen to proceed. Some metals, such as iron, may serve as a catalyst, increasing the rate of self-heating. The type of ignition and the temperature developed in a waste storage pile are dependent on such factors as:

- particle size;
- amount of organic material;
- moisture content;
- size of the waste pile;
- surface area of the waste fuel available to reaction; and
- the pressure over the pile (Tamaddon et al., 1995).

Fresh organic waste is sensitive to oxidation by biodegradation and presents a higher risk of spontaneous combustion than waste material that has already been exposed to the same biodegradation processes for a long period. The presence of carbon monoxide (CO) in the evolved gases is a good indicator of imminent self-ignition (Tamaddon et al., 1995).

Materials that increase the risk of spontaneous ignition in waste storage are food and garden waste, fats, dairy products, coal, plastics, sawdust and iron filings, jute fabric, sodium lamps, and waste from petrol stations such as oily rags and cans containing solvents (Thörnqvist, 1987).

Heat is produced in organic material when it is stored in large quantities, through a number of processes. Physical processes, such as adsorption of water to dry surfaces, or condensation of water vapour on a cold surface, can lead to heat production (Thornqvist, 1987), but these physical processes are more significant at low temperatures, i.e., 20 °C or less (Fig. 1.1).

Microbiological activity can also contribute to the temperature increases in the range 0–75 °C, but such activity is of greatest importance for temperature development between 20–60 °C. The unsorted industrial waste (IND) fuel storage pile investigated in **Paper I** contained some household and garden waste, rich in easily degradable carbohydrates. This resulted in a rapid temperature increase, so that, within a month of storage, the temperature increased from an average of 49 °C to 73 °C because of microbiological activity. A similar temperature increase has been observed during the storage of wood fuel (Thörnqvist, 1987).

Above 70 °C, there is still some microbiological activity from thermophilic bacteria. However, the dominant heat-generating processes at these higher temperatures are chemical oxidation processes (Fig. 1.1). Ernstson (1995) observed an optimal degradation rate between 25–40 °C, a slow rate at 15 °C and a negligible rate at 55 °C in controlled laboratory experiments with stored forest fuel materials.

Chemical oxidation liberates heat, but requires elevated temperatures. Chemical oxidation processes start to contribute to the temperature increase even at 40 °C. In the range 40–50 °C, about 20% of the increase in temperature is due to chemical processes, and between 50–60 °C, about 30% is due to chemical processes. Between 60–70 °C, chemical processes prevail over biological processes and can contribute up to 60% of the increase in temperature. In other words, microbiological processes create the temperature necessary for chemical processes to proceed. These elevated temperatures can then start an exothermic chemical reaction, which may lead to spontaneous combustion.

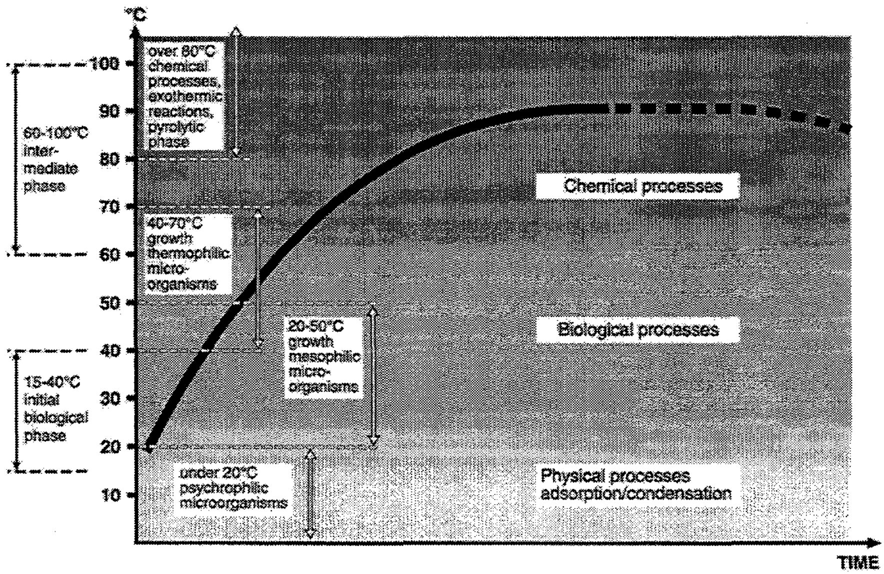


Figure 1.1 Contributions of physical, biological and chemical processes to the temperature profile in a waste fuel storage pile (**Paper I**).

Biodegradation reactions cannot occur if the water content in the material is less than 20–25% (Ernstson and Rasmuson, 1988). These authors emphasized that moisture content is an important parameter in the heating process and its influence is twofold. The temperature increase is lower in moist material because the heat capacity of the bed is higher. At higher temperatures, evaporation leads to a decrease in heating rate by removing heat efficiently at the same time as the moisture decreases locally. The transport of water is transient in nature, but changes in the moisture content take place much more slowly than changes in oxygen concentration and temperature (Ernstson and Rasmuson, 1990).

1.1.2 Theoretical basis for modelling oxygen, water and heat transport

Degradation of organic material consumes oxygen and produces heat, water and carbon dioxide. The relationships between different transport mechanisms and biological and chemical degradation in storages of forest fuel material have been mathematically modelled (Ernstson, 1995; Ernstson and Rasmuson, 1993). A sketch of the coupling between degradation and the transport of oxygen, moisture and heat is shown in Fig. 1.2. Free convection is produced when significant difference between the ambient temperature and that in the inner parts of the storage pile is established.

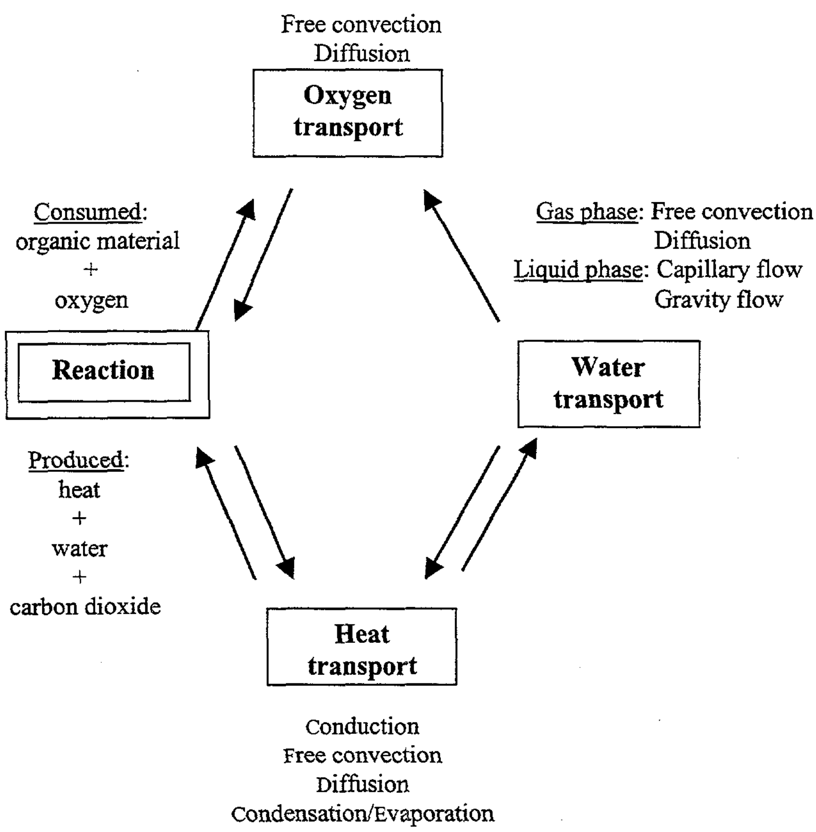


Figure 1.2 Diagram showing the coupling between waste degradation and transport of oxygen, moisture and heat (after Ernstson, 1995).

The following equations describing the flow of air, water, oxygen and heat transport were applied (Ernstson, 1995; Ernstson and Rasmuson, 1993) to formulate a mathematical model for degradation and transport processes in stored waste fuel materials.

Flow of air

An equation for conservation of mass (Eq. 1), an equation of motion (Darcy's law) of air (Eq. 2), and an equation that accounts for a linear change in fluid density, ρ, with temperature rise (Eq. 3).

$$\nabla \cdot \vec{q}_a = 0 \tag{1}$$

$$\vec{q}_a = -\frac{k}{\mu}\left(\nabla p - \rho \vec{g}\right) \tag{2}$$

$$\rho = \bar{\rho}\left[1 - \bar{\beta}\left(T - \bar{T}\right)\right] \tag{3}$$

where:
∇ stands for the mathematical grad operation (gradient of a scalar parameter) and

q_a = Darcy flow applied to the air flow, [m^3 m^{-2} s^{-1}]
k = permeability, [m^2]
μ = viscosity, [N s m^{-2}]
p = pressure, [N m^{-2}]
ρ = density, [kg m^{-3}]
g = gravitational acceleration, [m s^{-1}]
β = volume expansion coefficient, [K^{-1}]
T = temperature, [K]

Water transport in the liquid phase

An equation for the conservation of mass for liquid water (Eq. 4), and a corresponding equation of motion (Darcy's law for unsaturated porous media) (Eq. 5).

$$\rho_w = \frac{\partial \theta}{\partial t} = -\rho_w \nabla \cdot \vec{q}_w - \frac{\partial E}{\partial t} + F \tag{4}$$

$$\vec{q}_w = -K(\theta)\ \{\nabla \psi(\theta) + 1\} \tag{5}$$

where:
ρ_w = density of the water, [kg m^{-3}]
θ = moisture content, [m^3 water m^{-3} bed]
t = time, [s]
q_w = Darcy flow applied to the water flow, [m^3 m^{-2} s^{-1}]
E = mass concentration of evaporated water, [kg m^{-3}]
F = production rate of water, [kg m^{-3} s^{-1}]
$K(\theta)$ = parameter characterizing the porous material, [m s^{-1}]
$\psi(\theta)$ = parameter characterizing the flowing fluid, [m H_2O]

F is the amount of water produced and $\partial E/\partial t$ is the net amount of water evaporated in the degradation.

Water transport in the gas phase:

Equation for transport of water vapour Eq. (6). This includes transport of water vapour by both diffusion and convection and vapour produced by evaporation.

$$\varepsilon\ (1-\alpha)\frac{\partial C_v}{\partial t} = D_v \nabla^2 C_v - \vec{q}_a . \nabla C_v + \frac{1}{M}\frac{\partial E}{\partial t} \tag{6}$$

where:

ε = porosity
α = moisture content, [m^3 water m^{-3} void]
C_v = concentration in the gas (water vapour) phase, [mol m^{-3}]
q_a = Darcy flow applied to the air flow, [m^3 m^{-2} s^{-1}]
D_v = effective diffusion coefficient for water vapour, [m^2 s^{-1}]
M = molecular weight, [g mol^{-1}]

Oxygen transport

Equation 7 describes the transport of oxygen in the gas phase. It includes the effects of diffusion, convection and oxygen consumption, r_x, on the degradation.

$$\varepsilon\ (1-\alpha)\frac{\partial C_x}{\partial t} = D_x \nabla^2 C_x - \vec{q}_a \cdot \nabla C_x - r_x \tag{7}$$

where:

α = moisture content, [m^3 water m^{-3} void]
C_x = concentration in the gas (oxygen) phase, [mol m^{-3}]
D_x = effective diffusion coefficient for oxygen, [m^2 s^{-1}]
q_a = Darcy flow applied to the air flow, [m^3 m^{-2} s^{-1}]
r_x = first order reaction rate (oxygen consumption), [mol m^{-3} s^{-1}]

Heat transport

Equation 8 expresses the heat balance, including heat transported by conduction and convection.

$$\left(\bar{\rho}\ \bar{c}_p\right)\frac{\partial T}{\partial t} = \lambda\ \nabla^2 T - \left(\bar{\rho}\ \bar{c}_p\right)\ \vec{q}_a \cdot \nabla T + Q - r_v \frac{\partial E}{\partial t} \qquad (8)$$

where:

c_p = heat capacity at constant pressure, [J kg^{-1} °K^{-1}]
λ = thermal conductivity, [J s^{-1} m^{-1} °K^{-1}]
Q = heat production rate per unit volume due to degradation, [m^3 m^{-2} s^{-1}]
r_v = heat of evaporation [J g^{-1}], where g is gravitational acceleration [m s^2]

Here, the expression $r_v \dfrac{\partial E}{\partial t}$ is the heat needed for evaporation.

Due to the complexity involved in the overall problem of air flow and water, oxygen and heat transport, coupled with microbial degradation some simplifications and an approach by steps are suggested (Ernstson and Rasmuson, 1993 and Ernstson, 1995).

First, the airflow distribution due to natural convection is calculated, assuming a given temperature gradient. Secondly, one-dimensional oxygen and heat transport and waste degradation are estimated along single stream tubes. In these calculations, the moisture content is everywhere assumed constant and optimal for the degradation. This simplification is based on that changes in moisture content are much slower than the change in the oxygen concentration and temperature.

Simple water transport calculations are performed using the calculated temperature profiles and airflow rates. Finally, a simplified flow distribution is used in a two-dimensional oxygen, heat transport and degradation model.

The mathematical model, including oxygen and heat transport equations coupled with degradation reactions is solved by Ernstson (1995) with the computer code TWODEPEP, which uses the finite-element method for solving several simultaneous partial differential equations in two-dimensional and one-dimensional region. Detailed procedure is found in Ernstson and Rasmuson (1993) and Ernstson (1995).

Both one- and two-dimensional models used in conjunction with laboratory and field experiments with chipped forest fuel material indicated that natural convection plays a major role in the degradation process by supplying oxygen and releasing heat (Ernstson, 1995).

1.1.3 Natural convection

Natural convective movements depend on the density differences caused by the temperature gradients and on the boundary conditions. When the temperature within the pile rises, gas flow may be induced by buoyancy effects. The transport of oxygen, heat and water in the inner part of the pile is likely to occur mainly in the gas phase.

It has been found that during storage of wood fuels, steam transport by natural convection played an important role in heat transport, by comparison with thermal conduction. Furthermore, it was also shown that natural convection was the dominant mechanism for oxygen transport, rather than diffusion (Ernstson and Rasmuson, 1990; Ernstson and Rasmuson, 1993; and Ernstson, 1995).

The shape of the induced gas flow field can be estimated, to a large degree, by the temperature profile, permeability, and geometry of the pile (Collin et al., 1986). Air flows in through the sides of the heap, close to ground level, while outflow occurs in the upper, horizontal layer of the pile (Fig. 1.3). Flow never reaches the inner, lower part of the pile. The extent of the region in which airflow occurs is dependent on the temperature profile. Natural convection is of great importance in the upper part of the pile and close to the slope surface, while convective flow in the inner, lower part of the pile is very low.

Natural convection is a result of temperature increase in the storage piles where the heated, rising gas induces a corresponding suction. Gas flow rates can be estimated from the density differences created by the temperature gradient, and the boundary conditions of the pile. In order to be able to calculate the gas flow created by natural convection, the temperature profile and variation in gas permeability throughout the pile must be known.

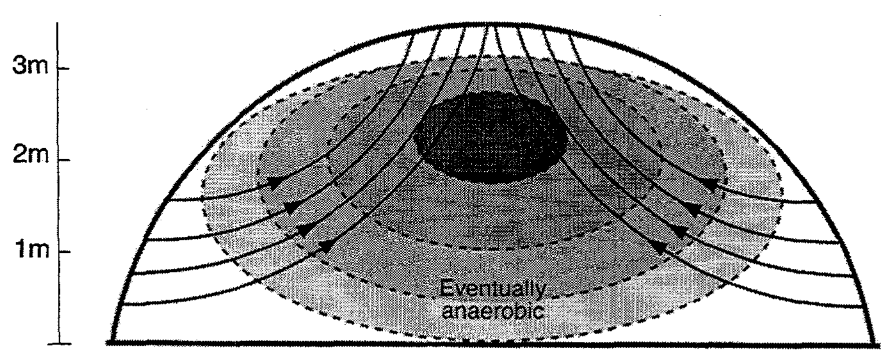


Figure 1.3 Cross section through a fuel material storage pile (after Collin et al., 1986). The grey gradient symbolizes different zones, which create gas flow by convection.

Permeability is a very important parameter in the theory of oxygenation of storage piles. For wood chip piles, permeability has been estimated and the information used to evaluate the importance of transport mechanisms for O_2, steam and heat. These three parameters are directly related to the decomposition rate.

In the study presented in **Paper I** permeability was not investigated. Since the incoming material (industrial waste IND and refused derived fuel RDF) is very heterogeneous, estimations made on one or a few samples would not have been representative of the whole material. The permeability of waste piles also changes locally during storage due to non-uniform compaction rates in different parts of the pile, which are related to non-uniform biodegradation rates due to variable content of readily degradable substances, and the weight of the material.

Transport processes identified in piles of forest fuel cannot be applied to explain what occurs in waste fuel storage piles without some degree of uncertainty. Waste fuel differs from wood chips, due for example, to the greater heterogeneity of the former with respect to particle size, differing moisture content, composition and degradation rates. Therefore, it is more difficult theoretically to describe the processes that occur during the storage of waste fuel. However, the theoretical basis established for forest fuel storage (Ernstson, 1995; Ernstson and Rasmuson, 1993; Ernstson et al., 1991; Ernstson and Rasmuson, 1990; Thörnqvist, 1988; Kubler, 1987; Collin et al., 1986; Collin et al., 1986) is, in many respects, applicable to the results described in **Paper I** regarding the storage of waste fuels.

1.2 Hydrological Performance of Landfills

The water balance method and analytical and numerical models are commonly used to predict the percolation rate beneath landfills (Guyonnet et al., 1998; Uguccioni and Zeiss, 1997; Zeiss and Uguccioni, 1995; Nolting et al., 1995; Schroeder et al., 1994; Bengtsson et al., 1994; Zeiss and Major, 1993; Blakey, 1992; Ettala, 1992; Kalqvist, 1987; Ehrig, 1983; Holmes, 1983). Most of these studies were performed on MSW landfills.

Recently, increased attention has been paid to environmental aspects related to landfill disposal of solid wastes other than MSW, such as the bottom and fly ashes from municipal solid waste incinerators, coal combustion ashes, sewage sludge, iron- and steel-making slags and asbestos (Cossu et al.,1997; Crawford et al.,1997; de Angelis and Balzamo, 1997; Gavasci et al., 1997; Heyer and Stegmann, 1997; Hjelmar and Birch, 1997; Lechner et al., 1997; Savvides et al., 1997; Hjelmar, 1995). An obvious problem with most methods of ash pre-treatment before landfilling is their high capital and operating costs (Nilsson, 1998). Therefore, a significant proportion of MSW incineration residues (MSWI) is likely to be landfilled without any pre-treatment.

Since water often carries and distributes pollutants throughout the environment, it is important from both short- and long-term perspectives to understand water-flow mechanisms in present and future landfills. Assessment of the hydraulic characteristics of landfills is a relevant requirement due to the potential impact on groundwater quality of uncontrolled leachate releases (Oweis, 1990).

1.2.1 Models for simulation of leachate production

Estimation of landfill leachate production in its simplest form is made using the water balance method, and many examples of its application are available (Nolting et al., 1995; Bengtsson et al., 1994; Blakey, 1992; Ehrig, 1983). Even though it is more practical, the water balance method does not always adequately reproduce observed patterns of landfill leachate production. The water balance equation can be presented as:

$$Q = P - R - ET - \Delta\theta \qquad (9)$$

where
Q, P, R, ET and θ represent leachate generation (discharge), precipitation, surface runoff, evapotranspiration and water storage, respectively.

The Hydrologic Evaluation of Landfill Performance (HELP) model (Schroeder et al., 1994) goes into more detail in trying to account for the complexity of the system than do most water balance models. By the time of its publication, HELP model Version 3 had been widely tested under different conditions (Schroeder et al. 1994) and, since then, has been one of the most widely used computer models in landfill hydrological studies. However, its applicability to modelling leachate discharge from uncovered landfills has not been clearly demonstrated.

The HELP model takes into account the accumulation of water up to field capacity and the time lag in the precipitation-leachate discharge relation. The HELP model is a quasi-two-dimensional (2D) deterministic model. It handles one-dimensional (1D) vertical drainage and 1D lateral drainage coupled at the base of lateral drainage layers on top of liner/barrier systems. Surface hydrological processes include snowmelt, interception of rainfall by vegetation, surface runoff and evaporation. Subsurface hydrological processes include: evaporation from a soil profile, plant transpiration, unsaturated vertical drainage, soil barrier/liner percolation, geomembrane leakage, and saturated lateral drainage.

The HELP model predicts leachate flow via a 1D uniform Darcian flow through a homogeneous, unsaturated porous medium with constant porosity, pore size distribution, bubbling pressure and residual saturation (Schroeder et al., 1994). Darcian flow requires that the flow be laminar and uniform. Although Darcy's Law was originally developed for saturated flow, it can be used for unsaturated conditions by expressing the hydraulic conductivity as a function of the suction head.

$$q = -K(\Psi)\Delta(h) \tag{10}$$

where:
q = specific discharge, [cm s^{-1}]
K = hydraulic conductivity, [cm s^{-1}]
Ψ = capillary pressure, [cm]
h = vertical thickness of the waste layer, [cm]

When the flow is considered to be one-dimensional and the medium is saturated, Eq. 10 simplifies to Darcy's law.

The unsaturated hydraulic conductivity (K_{us}) varies with soil moisture content. The adjustment of K_{us} is accomplished through application of the combined Brooks-Corey (Brooks and Corey, 1964) and Campbell (Campbell, 1974) equations, respectively, Eq. 11 and Eq. 12, which relate K and Ψ indirectly by making both a function of the moisture content of the porous matrix in Eq. 13.

$$\frac{\theta - \theta_r}{n - \theta_r} = \left(\frac{\Psi_b}{\Psi}\right)^{\lambda} \tag{11}$$

$$K = K_s \cdot \left(\frac{\theta}{\theta_s}\right)^{3+\frac{2}{\lambda}} \tag{12}$$

$$q = K_s \cdot \left[\frac{\theta - \theta_r}{n - \theta_r} \right]^{3+\frac{2}{\lambda}} \cdot \frac{dh}{dl} \tag{13}$$

where:

θ = soil moisture content, [dimensionless]
θ_s = soil moisture content at residual saturation, [dimensionless]
n = porosity, [dimensionless]
Ψ_b = bubbling pressure, minimum capillary pressure on the drainage cycle for which a continuous non-wetting (air) phase exists, [cm]
K_s = saturated hydraulic conductivity, [cm s^{-1}]
l = length in the direction of flow, [m]

Equation 2.5 represents moisture movement as vertical plug flow, since K_s, θ, n and λ are considered constant through the medium. The equation can be further simplified by setting the hydraulic gradient to unity (Uguccioni and Zeiss, 1997).

The effects of capillarity on the unsaturated flow rate are not considered in the HELP model. As a consequence, unsaturated drainage flow occurs only when the moisture content of a layer has reached field capacity, provided that no underlying layer has lower moisture content than the layer in question (Schroeder et al. 1994).

The HELP model also does not take channel flow into account. In the following cited literature references it was emphasized that a 2D flow regime of channelled and matrix flows may better describe the water flow inside a MSW landfill. The predicted breakthrough time given by HELP for MSW landfills is usually longer than that observed (Uguccioni and Zeiss, 1997).

Several investigations have indicated that flow channelling through waste layers plays an important role in discharge patterns (Bendz, 1998; Guyonnet et al.,1998; Uguccioni and Zeiss, 1997; Zeiss and Uguccioni, 1995; Zeiss and Major, 1993; Blight et al., 1992; Stegmann and Ehrig, 1989). Early appearance of leachate before the attainment of field capacity has been reported (Stegmann and Ehrig, 1989).

The proportion of infiltration into the waste that percolates rapidly through preferential pathways has been estimated and calibrated (Guyonnet et al., 1998; Uguccioni and Zeiss, 1997; Zeiss and Uguccioni, 1995; Zeiss and Major ,1993; Blight et al., 1992; Stegmann and Ehrig, 1989). The channelling effect seems to be mainly related to high infiltration rates and is most significant in young deposits due to their coarser structure.

A modified landfill water balance model named MOBYDEC was developed (Guyonnet et al., 1998). With respect to infiltration through the cover material, the model is based on the classical water balance approach. The less familiar features are related to the manner in which the evolution of the water content of the waste is handled. According to the authors, the model avoids the use of a waste hydraulic conductivity, as the applicability of this medium property is questionable for MSW due to its non-Darcian nature.

The MOBYDEC model assumes that the waste behaves as a "double porosity system". Leachate peaks that are correlated in time with rainfall events can be reproduced by allowing a certain proportion of net infiltration into the waste to percolate rapidly through preferential pathways. The remaining net infiltration is absorbed by the waste and released according to a first-order kinetic relationship:

$$\frac{\partial \theta}{\partial t} = -\frac{\ln 2}{T_{0.5}} \cdot (\theta - \theta_c) \qquad \text{if} \quad \theta \geq \theta_c \tag{14}$$

where:
θ = water content of the waste at time t, [mm]
θ_c = water content at which leachate starts to appear (referred to as the critical water content), [mm]
$T_{0.5}$ = "depletion half-life".

The "depletion half-life" $T_{0.5}$ is related to the speed at which absorbed water is released by the waste and appears in the leachate collection system. It is the time required for the volume of absorbed water to reduce to half of its initial value.

The proportion of infiltration that percolates rapidly through preferential pathways was estimated at between 1% and 23% when six full-scale cells in two landfill sites were studied (Guyonnet et al., 1998). According to those authors, the proposed MOBYDEC model is oriented towards calibration with measured data and intermediate term predictions. The model is not believed to be appropriate for long-term predictions of leachate production. For such a purpose, a more "mechanistic" approach would be required.

A more physically-sound dual-domain flow model for a fractured porous medium (PREFLO) that takes channelling into account has been calibrated to MSW (Ugoccioni and Zeiss, 1997). In this model, the flow through the matrix is calculated by Richard's equation with sink terms for water removal by roots, addition of water from channels, and removal or addition of water from boundaries. When the water input exceeds the saturated hydraulic conductivity of the matrix, the excess water flows down the channels according to Poiseuille's Law. Lateral flow exchange between a channel and the matrix obeys Darcy's Law.

Despite the physical superiority of PREFLO over HELP, the specification of parameters such as channel diameter, length, and others, required by PREFLO is difficult on a laboratory scale and even harder to define at field scale.

One method of implicitly accounting for channelling when simulating leachate generation by the HELP model has been demonstrated (Uguccioni and Zeiss, 1997; Zeiss and Uguccioni, 1995; Zeiss and Major, 1993). By modelling leachate generation in MSW test cells it was found that the experimental unsaturated hydraulic conductivity was four to five orders of magnitude higher than the HELP default value at field capacity (FC) (1.2×10^{-7} cm/s) (Zeiss and Uguccioni, 1995). By replacing the default value at FC by a "practical field capacity" of 0.136, which is much lower than the HELP default FC (0.292), the initial breakthrough time could be reproduced (Uguccioni and Zeiss, 1997).

It has been emphasized that in landfill leachate modelling, as in most modelling, the complexity of the selected model should be consistent with the detailed data that is available or that can be expected (Guyonnet et al., 1998). Therefore, the selection of an appropriate model should be made on a case-by-case basis.

1.3 Stormwater Runoff and Pollutant Transport

Stormwater runoff refers to the volumes and rates of water flow in individual storm events. It is often called *direct runoff* because it results mainly from surface flow and other immediate responses to precipitation (Ferguson, 1998). The first step in estimating pollutant loads transported by stormwater, is to establish their magnitude.

A runoff gauging station would provide a direct, factual way to observe flows from a site in its existing condition. However, very few waste management parks or landfill sites have such a gauging station. One reason for this is that most of the waste storage and waste treatment activities are carried out on surfaces surrounded by infiltration ditches connected to drainage systems into which other types of water are drained. At these sites, infiltration is the main strategy for handling stormwater runoff.

Therefore, some sort of estimate is necessary, based on data about the site and general knowledge of runoff processes. According to the modelling principle of "parsimony", a simpler model should be preferred over more complex models, particularly where assessment of model "validity" cannot easily be carried out, as was the case for the majority of areas and roads at the Spillepeng site described in **Paper III**.

1.3.1 Return period

Rainfall runoff models estimate runoff on the basis of drainage area and the rainfall upon it. The magnitude of a storm expressed either as an intensity (e.g., mm per hour) or as total amount during the whole event (e.g., total mm in a one-hour storm) has a *return period*, that is, the average period in which a storm of that type will re-occur.

An extreme event is defined to have occurred if a random variable X is greater than or equal to a level x_T. The *recurrence interval*, τ, is the time between occurrences of $X \geq x_T$. In the Malmö region for instance, based on a 30-year-period of registered rainfall events (Fig. 1.4), if x_T = 36 mm, there are seven *recurrence intervals*, τ, covering the total period of 30 years between the first and the last event $X \geq x_T$.

The *return period*, *T*, of the event $X \geq 36$ mm is the average value, $\bar{\tau}$, measured over a large number of occurrences. For the Alnarp-Frukt rainfall data series, the *return period* of an event $X \geq 36$ mm is the average $\bar{\tau} = 30/7 = 4.3$ years.

The probability, *p*, of occurrence of an event $X \geq x_T$ in any observation may be related to the *return period* as follows.

For each observation, there are two possible outcomes: either success $X \geq x_T$ (probability p), or failure $X < x_T$ (probability 1- p). Since the observations are independent, the probability of a *recurrence interval* of duration τ is the product of the probabilities of τ - 1 failures followed by one success, that is, $(1-p)^{\tau-1}p$.
The expected value of τ is given by:

$$E(\tau)=\sum_{\tau=1}^{\infty}\tau\ (1-p)^{\tau-1}\ p \tag{15}$$

This equation may be written as:

$$E(\tau)=\frac{p}{[1-(1-p)]^2}=\frac{1}{p} \tag{16}$$

Hence, E(τ) = T = $1/p$; which means that the probability of the occurrence of an event in any observation is the inverse of its *return period*.

$$p(X \geq x_T)=\frac{1}{T} \tag{17}$$

In the previous example, the *return period* is T = 4.3 years, and p = 0.23.

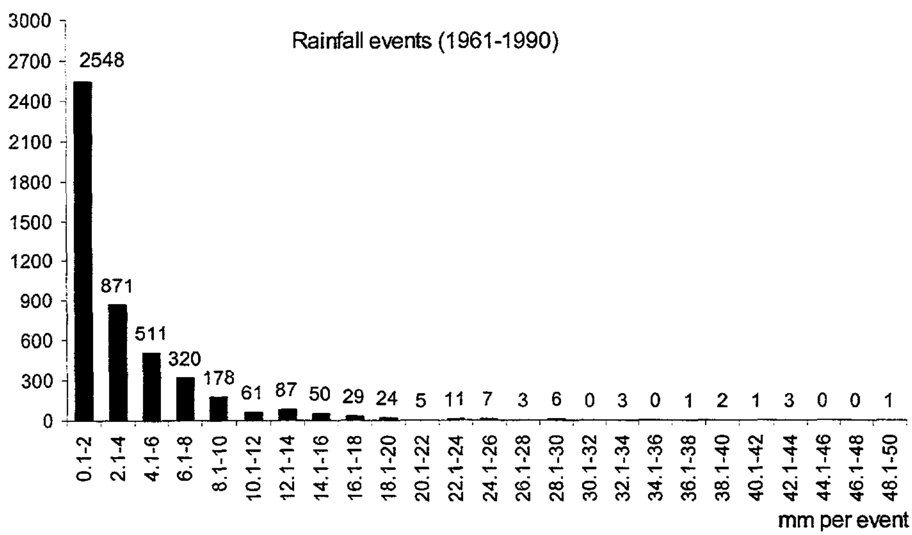


Figure 1.4 Rainfall events in depth (mm). Alnarp-Frukt station, Malmö region, southern Sweden, period 1961-1990.

1.3.2 Drainage area

The drainage area is the land area that drains to the point at which runoff is estimated. Any rainfall-runoff model requires clear identification of the drainage area concerning its size, soil and condition.

1.3.3 Time of concentration

Time of concentration is the maximum amount of time that runoff from any point in a drainage area takes to flow to the outlet from that area. Time of concentration is the longest possible time, whether or not it involves the longest distance (Ferguson, 1998). By definition, to calculate the time of concentration, an outlet must be associated with the area under investigation. The time of concentration can be calculated by using the empirical U.S. Federal Aviation Agency equation (FAA, 1965):

$$t_c = \left[1.8(1.1-c)L_h^{1/2}\right]/G^{1/3} \tag{18}$$

where:
t_c = time of concentration [min]
c = runoff coefficient [dimensionless]
L_h = hydraulic length [m]
G = slope along the hydraulic length [%]

1.3.4 Abstraction and runoff coefficients

In order to calculate stormwater runoff, the excess rainfall must be estimated, that is, the rainfall that is neither retained on the land surface nor infiltrates into the soil. After flowing across the surface, excess rainfall becomes direct runoff under the assumption of Hortonian overland flow (Chow et al., 1988). The difference between the observed total rainfall and the excess rainfall is termed abstraction, or loss.

Abstraction includes:

a) storage in depressions on the ground surface as water accumulates in hollows;
b) interception of rainfall by vegetation above the ground;
c) infiltration into the soil.

Calculations for impervious surfaces and bare soil with low permeability (for instance, compacted clay) exclude abstraction of types b) and c). The losses in these cases are a consequence of depression storage in irregularities on the surfaces, and depend on the type of material used and the age of the road. On an asphalt surface, the depression storage is usually in the range of 0.5-1.5 mm (Hogland, 1986). Abstraction may also be accounted for by means of runoff coefficients. The most common definition of a runoff coefficient is the ratio of the peak rate of direct runoff to the average intensity of rainfall in a storm (Chow et al., 1988). Because of the high variability in rainfall intensities, this value is difficult to determine from observed data. A runoff coefficient can also be defined as the ratio of runoff to the rainfall over a given area and time period. These coefficients are most commonly applied to individual storm rainfall and runoff events, but can also be used for monthly or annual rainfall and stream flow data. If the total rainfall is:

$$\sum_{m=1}^{M} P_m \qquad (19)$$

where M is the number of intervals and P_m is the observed depth of rainfall in time interval m; and denoting Q_d as the corresponding depth of runoff, a runoff coefficient, c, can be defined as:

$$c = \frac{Q_d}{\sum_{m=1}^{M} P_m} \qquad (20)$$

The degree of surface imperviousness is one of the most important characteristics of a surface, determining stormwater runoff generation. Impervious surfaces, such as asphalt roads, pavements and the roofs of buildings, will produce nearly 100% runoff after the surface has become thoroughly wet, irregardless of the slope. For non-paved surfaces, more complex processes take part in the generation of surface runoff. The infiltration rate decreases as rainfall continues, and is also influenced by the initial moisture content of the soil.

Other factors influencing the runoff coefficient for non-paved surfaces are ground slope, rainfall intensity, proximity of the water table, degree of soil compaction, porosity of the subsoil and vegetation. A reasonable coefficient must be chosen to represent the integrated effects of all these factors (Chow et al. 1988). The literature contains many reports of standard runoff coefficients for various agricultural and engineering applications.

1.3.5 The Rational method

The Rational method is about 100 years old (Kuichling, 1889). The Rational method is still widely used for stormwater design because of its simplicity. It was originally developed to estimate peak flows from small urban drainage areas, and its application today is usually restricted to drainage areas of less than 81 ha.

The Rational method associates peak rate of runoff with three easily identifiable characteristics of a drainage area and the rainfall upon it. The Rational formula assumes that, for a given *recurrence interval*, a storm duration that matches a drainage area's time of concentration produces the greatest rate of runoff. The equation is:

$$Q_p = A_d\, ci \tag{21}$$

where:

Q_p= runoff peak [m^3/h]

A_d = drainage area [m^2]

c = runoff coefficient, based on a combination of soil, land use, and slope

i = rainfall intensity at a selected *recurrence interval* and duration [mm/h]

When the Rational method was originally developed, it was intended only to identify the peak rates of flow that pipes and culverts had to carry. It translates peak intensity of rainfall directly into peak intensity of runoff. The method was not originally intended to estimate total volume of flow during a storm. For applications that involve volume of flow, the SCS method is preferable.

1.3.6 The SCS method

The development of the SCS method began during the 1950s by the U.S. Soil Conservation Service (SCS), which had its name changed in 1994 to Natural Resources Conservation Service (NRCS). Over the years, SCS has published a number of handbooks, updating the charts and step sequences, and the most useful one was published in 1986 (SCS, 1986).

Unlike the Rational formula, the SCS method begins by finding the total depth and volume of runoff during a storm. The rate of runoff in m^3/s is then determined by the distribution of rainfall intensity during the storm and how fast the given volume of runoff drains off the watershed. The basic equation of the SCS method is:

$$Q_d = \frac{(P - I_a)^2}{(P - I_a + S)} \tag{22}$$

where:

Q_d = depth of runoff [mm]

P = depth of 24-hour rainfall [mm]

I_a = initial abstraction [mm]

S = potential maximum retention after runoff begins [mm]

Initial abstraction, I_a, is defined as the losses of rainfall to infiltration and surface depressions before runoff begins (mm). The approximate median value found from field observations is $I_a = 0.2S$ (Ferguson, 1998).

Potential maximum retention, S, is defined by curve number CN, which is a function of the drainage area's soil and land use: $S = (1000/CN) - 10$

For precipitation, P, the SCS method always uses 24-hour rainfall, with the intensity of rain within that period assumed to be within a certain distribution.

The SCS method makes powerful use of the U.S. Soil Conservation Service's system of classifying soils and, therefore, its application is particularly suitable for U.S. soil conditions. The curve number, *CN*, is derived from a combination of the Hydrologic Soil Group (HSG) and land use (see Table 2-2 in U.S. SCS, 1986).
In a simplified way, the soils are classified basically into four groups (*A-D*) according to their potential of runoff generation.

Soils in group *A* have high infiltration rates and consist chiefly of deep, well-to-excessively drained sands or gravels.
Soils in group *B* have moderate infiltration rates when thoroughly wetted. This group includes silt loam, and loam that has experienced urbanization.
Soils in group *C* have low infiltration rates when thoroughly wetted and include sandy clay loam that has experienced urbanization.
Soils in group *D* have high runoff potential. They have very low infiltration rates when thoroughly wetted. This group consists of clay soils with high swelling potential, soils with permanent high water tables, and shallow soils over impervious material. This group also includes clay loam, silty clay loam, sandy clay, silty clay and clay that has experienced urbanization.

The curve number may vary for instance from $CN = 30$ (soil belonging to the hydrologic group *A*, with meadow/grass mowed for hay, not grazed in agricultural area) to $CN = 98$ (soil belonging to the hydrologic soil group D, with impervious pavements and roofs in urban areas).
Urban areas, which consist of combinations of pervious and impervious surfaces, involve construction of a special kind of composite *CN*.

Ferguson (1996) provided a simple method for estimating monthly direct depth runoff. It closely replicates the runoff that would be obtained by applying the SCS method to daily precipitation each day during the month and summing the results.

$$Q_d = 0.208\, P / S^{0.66} - 0.095 \tag{23}$$

A simplification of the SCS method that gives the runoff amount by volume, and which has been used in urban studies in Sweden uses the following equation:

$$Q_v = a \cdot A \cdot (P\text{-}S) \cdot 10^{-3} \tag{24}$$

where:
Q_v = runoff volume [m^3]
a = fraction of the impervious area that effectively contributes to runoff [dimensionless]
A = total impervious area [m^2]
S_D = depression storage [mm]

1.3.7 Estimation of stormwater pollutant load and sampling limitations

One method commonly used for estimating stormwater pollutant loads is based on the estimation of Event Mean Concentrations (*EMCs*). The method assumes that constituent concentration is constant throughout the simulation and the calculated load depends only on how well the *EMC* is estimated and the storm flows are measured (Charbeneau and Barrett, 1998). The *EMC* is defined as total constituent mass, *M,* discharged during an event, divided by the total volume, *V,* of discharge during that event (Huber, 1993). Mathematically:

$$EMC = \frac{M}{V} = \frac{\int C(t)Q_v(t)dt}{\int Q_v(t)dt} \qquad (26)$$

where:

$C(t)$ = constituent concentration at time t [mg L^{-1}]
$Q_V(t)$ = volume of stormwater discharge at time t [m^3]

The *EMC* is a flow-weighted average of constituent concentrations and is reported in units of mg L^{-1}. The total mass loading of a constituent during the storm may be obtained by multiplying *EMC* by the total volume of storm runoff. Such a model is useful for calculating annual loads when runoff coefficients and *EMCs* are estimated from monitoring of a number of storm runoff events. The use of *EMCs* is also appropriate for evaluation of the effects of stormwater runoff on receiving waters.

Receiving waters respond relatively slowly to storm inflows compared with the rate at which constituent concentrations change during a runoff event. Thus, the total load or *EMC* is the important parameter. If the *EMC* for a monitored event is used in a model, the total storm load will be adequately simulated, irregardless of how the actual constituent concentration changes during the runoff event. Field data show that *EMC* varies from one storm to another, and follows a log-normal distribution (Driscoll, 1986; U.S. EPA, 1983). It has been suggested that the use of an average *EMC* in multiple-event computer simulations may lead to estimates of total load that are as accurate as those obtained from more complex models (Charbeneau and Barrett, 1998).

To use the *EMC* method, special stormwater discharge monitoring points are necessary in order to install the flow-weighted sampler. At a waste management park with a stormwater drainage system such as at Spillepeng, which is based mainly on infiltration ditches, such outlet points are not available for each area. The stormwater accumulated in the ditches before infiltration, as well as impoundments formed during storm events, were the only sources available for sampling. In this case, the sampling points were needed to cover as much as possible of the area to avoid local effects on runoff composition. The composite sample obtained in this way was assumed with some uncertainty to represent the concentration of pollutants.

1.4 Groundwater Quality Time Series: Statistical Approach

A time series is a set of observations recorded over a period of time. Examples of time series include monthly rainfall data and daily river flow data. It is difficult to formulate conclusions based only on direct inspection of time series data, because of, for example, seasonal effects and random fluctuations. An appropriate statistical analysis is required.

Traditionally, analysis of such time series data sets have been carried out by selecting suitable parametric models such as autoregressive or moving average processes and then drawing inference from such series (Rao, 1999).

A time series has also been described as a mixture of the following possible components (Kendall, 1973):
(a) a trend, or long-term movement;
(b) fluctuations about the trend of greater or lesser regularity;
(c) a seasonal component; and
(d) a residual irregular, or random, effect.

It may be convenient to represent the series as the sum of these components or to give them a multiplicative association. One of the objects of time series analysis may be to break the series down into constituent parts for individual study. However, in doing so, we are, in effect, imposing a model that must be discarded if the model fails to fit the data. A trend can be defined as a smooth, broad movement of a non-oscillatory kind extending over a considerable period of time (Kendall, 1973). However, what appears as a climatic trend to a hydrologist may be nothing more than a temporary excursion or a short-term swing to a geologist, whose time-scale of interest is very much longer.

Concerning water quality time series, the detection and estimation of trends may be complicated by both particular data characteristics and the effects of sampling/ analysis procedures.
The main characteristics of water quality time series data sets are:
(a) seasonality;
(b) series correlation;
(c) lack of normality and skewness in distribution;
(d) flow-correlated quality; and
(e) censored data (i.e., data below analytical detection limits).

Common problems in water quality studies relating to sampling/analysis procedures, particularly for groundwater pollution monitoring, are:
(a) changes in sampling or analytical procedures that occur during long-term studies;
(b) effect of small sample size;
(c) time intervals between sampling not being equal;
(d) missing data;

(e) selection of inappropriate parameters for the aquifer/setting; and
(f) a lack of historical information on up-gradient groundwater pollution.

In many monitoring programmes, the existing data are often inadequate for statistical trend assessment, and the main problem is how to use the existing data sets, imperfect as they may be, to evaluate long-term trends.

A basic objective of a groundwater quality-monitoring programme for landfill sites is to detect pollution in time to allow deployment of measures to stop or at least attenuate the causes of the pollution. Usually such programmes aim to detect differences between up- and down-gradient locations and identify medium- and long-term trends.

Paper V addresses the extent to which these goals can be achieved using basic non-parametric statistics. The main objectives were to:
(a) detect and estimate changes occurring in the quality of the groundwater in a deep aquifer beneath the landfill site;
(b) assess the adequacy of various statistical methods and the assumptions made about the adequacy of quarterly sampling;
(c) verify the adequacy of the presently selected parameters and observation wells for the case study monitoring programme (Spillepeng landfill); and
(d) suggest improvements in the monitoring programme based on the results obtained.

The statistical procedures described below are related to these objectives. Emphasis was given to non-parametric tests of trend, since they are frequently considered most appropriate for water quality time series analysis (e.g., Harned et al., 1981; Gilbert, 1987; Loftis et al., 1991; Esterby et al., 1992; Helsel and Hirsch, 1992; Yu et al., 1993; Yu and Zou, 1993; McBride and Smith, 1997; Harned and Rao, 1998).

1.4.1 Comparison between groups

When the objective is to test significance of differences between two groups of data, such as the concentration of a groundwater pollutant, both up- and down-gradient of a landfill, selection of the statistical approach depends on the independence of the data sets and whether their distribution is normal or not (Table 1.1).

Such data sets are considered independent in the sense that there is not the same order of observations between sets. If there is a logical pairing of the observations between sets (i.e., observation 1 in set 1 is paired with observation 1 in set 2) then matched-paired tests are appropriate. A requirement to use matched-paired tests is that both sets must have the same number of measurements (same sample size). To compare sets of different sample size, methods for independent sets are indicated.

The methods for estimating the magnitude of the difference between the two sets are summarized in Table 1.

The sign test determines whether x is generally larger or smaller, or different, to y, regardless of whether that difference is additive. The sign test may be used regardless of the distribution of the differences, and is thus fully non-parametric.

The signed-rank test, sometimes called the Wilcoxon signed-rank test, is used to determine whether the median difference between paired observations equals zero. It may be used to test whether the median of a single data group is significantly different from zero.

Table 1.1 Tests to assess the significance of differences between data groups and (estimator of the magnitude of the difference between the two groups)

	Independent groups test	Matched-paired tests
Parametric test	t-test	Paired t-test
(Difference between means)	$(\bar{x}-\bar{y})$	$(\bar{x}-\bar{y})$
Non-parametric test		Sign test
(Difference between medians)		(D)
Non-parametric test	Rank-sum test	Signed-rank test
(Hodges-Lehmann estimator)	$(\hat{\Delta})$	$(\hat{\Delta})$

When the differences are symmetric, or in other words, x and y come from the same shaped distribution, differing only in the central value median (i.e., an additive difference), the signed-rank test has greater power to detect differences than does the sign test.

1.4.1.1 The rank-sum test for independent groups

In its most general form, the rank-sum test determines whether one group tends to produce larger values than the other group (Helsel and Hirsch, 1992). Usually, however, the test is used for a more specific purpose – to determine whether the two groups come from the same population (same median and other percentiles) or alternatively whether they differ only in location (median).

Grouping data from one set as x, and data from a second set as y, the null hypothesis H_0 states that the probability of any x value being higher than any given y value is one-half (0.5). The alternative hypotheses to H_0 are:

H_0: Prob. $[\, x > y \,] = 0.5$.

H_1: Prob. $[\, x > y \,] \neq 0.5$ (two-tailed test: x is expected to be larger or smaller than y).
H_2: Prob. $[\, x > y \,] > 0.5$ (one-tailed test: x is expected to be larger than y).
H_3: Prob. $[\, x > y \,] < 0.5$ (one-tailed test: x is expected to be smaller than y).

The exact test for independent groups

The exact test is appropriate for comparing groups of sample size 10 or smaller per group. The sample size for the smaller of the two groups x_i, $i = 1,\ldots..n$ is designated n, while the larger sample size y_j, $j = 1,\ldots m$ is designated m.

Compute the joint ranks $R_k = 1$ to $(N = n + m)$, using average ranks in case of ties.

W_{rs} = sum of ranks for the group having the smaller sample size.
$W_{rs} = \sum Ri \;\; i$=1,n (either group may be used when sample sizes are equal, i.e., $n = m$).

Reject H_0: Prob. $[\, x > y \,] = 0.5$ in favour of one of the alternative hypotheses:

H_1: Prob. $[\, x > y \,] \neq 0.5$ Reject H_0 if $W_{rs} \leq x^*_{\alpha/2,n,m}$ or $W_{rs} \geq x_{\alpha/2,n,m}$
H_2: Prob. $[\, x > y \,] > 0.5$ Reject H_0 if $W_{rs} \geq x_{\alpha,n,m}$
H_3: Prob. $[\, y > x \,] < 0.5$ Reject H_0 if $W_{rs} \leq x^*_{\alpha,n,m}$

The table for quantiles (p-values) for the rank-sum test statistic W_{rs} is used (for instance, Table B4, Appendix B in Helsel and Hirsch, 1992).

The large-sample approximation for independent groups

When both sets have sample sizes greater than 10 (n, m > 10), the large-sample approximation may be used. In this case, the distribution of the test statistic W_{rs} closely approximates a normal distribution. This approximation does not imply that the data are, or must be, normally distributed. Only the modified test statistic W_{rs} follows a normal distribution.

If there are no ties, W_{rs} has a mean μ_W and standard deviation σ_W when H_0 is true:

$$\mu_W = n \cdot (N+1)/2 \tag{27}$$

$$\sigma_W = \sqrt{n \cdot m \cdot (N+1)/12} \tag{28}$$

where $N = n + m$

Z_{rs}, the standardized form of the test statistic, is therefore computed as:

$$Z_{rs} = \begin{cases} \dfrac{W_{rs} - \dfrac{d}{2} - \mu_W}{\sigma_W} & if \quad W_{rs} > \mu_W \\ 0 \quad if \quad W_{rs} = \mu_W \\ \dfrac{W_{rs} + \dfrac{d}{2} - \mu_W}{\sigma_W} & if \quad W_{rs} < \mu_W \end{cases} \tag{29}$$

Z_{rs} is compared with a table of the standard normal distribution for test evaluation.

Correction for ties

A correction to σ_W when ties occur and tied ranks are assigned suggested by Conover (1980) is suitable for computing the large sample approximation.

The standard deviation is:

$$\sigma_{Wt} = \sqrt{\frac{n \cdot m}{N \cdot (N-1)} \cdot \sum_{k=1}^{N} R_k^2 - \frac{n \cdot m \cdot (N+1)^2}{4 \cdot (N-1)}} \tag{30}$$

where $N = n + m$

1.4.1.2 Matched-pair tests for paired groups: The sign-test and the signed-rank test

For paired observations (x_i, y_i), $i = 1, 2, \ldots n$, their differences $D_i = x_i - y_i$ are computed. The matched-pair tests determine whether x_i and y_i are from the same population (H_0) by analysing the D_i. When the D_i's have a normal distribution, a pai red t-test can be employed. The paired-t test determines whether the mean of the D_i's equals 0, which is equivalent to stating that the mean of the x_i and the y_i are the same. If the D_i's are symmetric, but not necessarily normal, a signed-rank test can be used. If the differences are asymmetric, the sign test may be used.

a) The sign test

The sign test determines whether x is generally larger (or smaller, or different) than y, without regard to whether that difference is additive (see definition in 1.4.1.2.2 Signed-Rank Test). It may be used regardless of the distribution of the differences, and thus is fully non-parametric.

The null and alternative hypotheses are:
H_0: Prob. $[\, x > y \,] = 0.5$
H_1: Prob. $[\, x > y \,] \neq 0.5$ (two-tailed test: x is expected to be larger or smaller than y).
H_2: Prob. $[\, x > y \,] > 0.5$ (one-tailed test: x is expected to be larger than y).

H_3: Prob. $[\, x > y \,] < 0.5$ (one-tailed test: x is expected to be smaller than y).

<u>The exact form of the sign test</u>

Tied data pairs (all $D_i = 0$) are ignored. The sample size of the test is reduced to the number of non-zero differences n. Assign a + for all $D_i > 0$, and a – for all $D_i < 0$.

The test statistic S^+ is the number of +'s, the number of times $x_i > y_i$, $i = 1, \ldots n$.

Reject H_0: Prob. $[\, x > y \,] = 0.5$ in favour of one of the alternative hypothesis:

H_1: Prob. $[\, x > y] \neq 0.5$ Reject H_0 if $S^+ \leq x'_{\alpha/2,n}$ or $S^+ \geq x_{\alpha/2,n}$
H_2: Prob. $[\, x > y \,] > 0.5$ Reject H_0 if $S^+ \geq x_{\alpha,n}$
H_3: Prob. $[\, x > y \,] < 0.5$ Reject H_0 if $S^+ \leq x'_{\alpha,n}$

The table for quantiles (p-values) for the rank-sum test statistic S^+ is used (e.g., Table B5, Appendix B in Helsel and Hirsch, 1992).

<u>The large-sample approximation for the sign test</u>

For sampling data set sizes of $n > 20$, the exact sign test statistic can be modified so that its distribution more closely follows a standard normal distribution.

The large sample approximation for the sign test takes the standardized form:

$$Z^+ = \begin{cases} \dfrac{S^+ - \frac{1}{2} - \mu_{S^+}}{\sigma_{S^+}} & if \quad S^+ > \mu_{S^+} \\ 0 \quad if \quad S^+ = \mu_{S^+} & \\ \dfrac{S^+ + \frac{1}{2} - \mu_{S^+}}{\sigma_{S^+}} & if \quad S^+ < \mu_{S^+} \end{cases} \tag{31}$$

where $\mu_{S^+} = \dfrac{n}{2}$, and $\sigma_{S^+} = \dfrac{1}{2}\sqrt{n}$

Z^+ is compared with a Z table for the standard normal distribution to obtain the approximate p-value.

b) The signed-rank (Wilcoxon) test

The signed-rank test determines whether the median difference between paired observations (median of the D_i's) is equal to 0.

This test is usually described as a determination of whether the x's and y's come from the same population (same median and other percentiles) or alternatively, that they differ only in location (median). If they differ in location, the differences will be symmetric when x and y come from data sets with the same shaped distribution (whatever the shape), differing only in their medians. This is called an *additive* difference between two sets, meaning that the variability and skewness within each group is the same for both. For additive differences, the signed-rank test has more power than does the sign test.

The signed-rank test is also appropriate when the differences are not symmetric in the units being used, but a logarithmic transformation of both data sets will produce differences that are symmetric. In this case a *multiplicative* relationship is made into an additive relationship in the logarithms (Helsel and Hirsch, 1992).

When the signed-rank test is performed on asymmetric differences, it rejects H_0 slightly more often than it should (Helsel and Hirsch, 1992).

The exact form of the signed-rank test

For $D_i = x_i - y_i$, the null hypothesis for the signed-rank test is stated as:

Reject H_0: median $[D] = 0$ in favour of one of the following alternative hypotheses:

H_1: median $[D] \neq 0$ (2-tailed test – x might be larger or smaller than y).
H_2: median $[D] > 0$ (1-tailed test – x is expected to be larger than y).
H_3: median $[D] < 0$ (1-tailed test –x is expected to be smaller than y).

Compute the absolute value of the differences $|D_i|$, $i = 1 \ldots N$. Rank the $|D_i|$ from smallest to largest. The test uses only non-zero differences, so sample size $n = N -$ [number of $D_i = 0$]. Compute the signed rank R_i, $i = 1, \ldots n$.

$$R_i = \begin{cases} rank\ of\ |D_i|\ for\ D_i > 0 \\ -(rank\ of\ |D_i|)\ for\ D < 0 \end{cases} \tag{32}$$

If $D_i = 0$, delete. When two non-zero differences D_i's are tied, assign the average of the ranks involved to all tied values.

The exact test statistic W^+ is then the sum of all signed ranks R_i having a positive sign:

$$W^+ = \sum_{i=1}^{n} (R_i | R_i > 0) \quad \text{where | signifies ‘given that’.} \tag{33}$$

Reject H_0: median $[D] = 0$ in favour of one of the alternative hypotheses:

H$_1$: median $[D] \neq 0$ Reject H$_0$ if $W^+ \leq x'_{\alpha/2,n}$ or $W^+ \geq x_{\alpha/2,n}$
H$_2$: median $[D] > 0$ Reject H$_0$ if $W^+ \geq x_{\alpha,n}$
H$_3$: median $[D] < 0$ Reject H$_0$ if $W^+ \leq x'_{\alpha,n}$

A table of quantiles (p-values) for the signed-rank statistic W^+ is used (for instance, Table B6, Appendix B in Helsel and Hirsch, 1992)

<u>The large-sample approximation for the signed-rank test</u>

For samples sizes $n > 15$, to avoid the requirement for a large table of exact signed-rank test statistics for all possible sample sizes, the exact test statistic is standardized by subtracting its mean and dividing by its standard deviation so that its distribution more closely follows a standard normal distribution.

The large sample approximation for the signed-ranks test takes the standardized form:

$$Z_{sr^+} = \begin{cases} \dfrac{W^+ - \frac{1}{2} - \mu_{W^+}}{\sigma_W} & if \quad W^+ > \mu_{W^+} \\ 0 \quad if \quad W^+ = \mu_{W^+} & \\ \dfrac{W^+ + \frac{1}{2} - \mu_{W^+}}{\sigma_{W^+}} & if \quad W^+ < \mu_{W^+} \end{cases} \tag{34}$$

where:

$$\mu_{W^+} = \frac{n \cdot (n+1)}{4} \quad \text{and} \quad \sigma_{W^+} = \sqrt{\frac{n \cdot (n+1) \cdot (2n+1)}{24}}$$

$Z_{sr}+$ is compared with a Z table for the standard normal distribution to obtain the approximate p-value for the signed-rank test.

1.4.2 The magnitudes of differences and confidence intervals

1.4.2.1 The Hodges-Lehmann estimators $\hat{\Delta}$ for rank-sum test

One non-parametric estimation method for determining the magnitude of the difference between two independent groups or two paired groups is the Hodges-Lehmann estimators $\hat{\Delta}$ (Hodges and Lehmann, 1963). These are computed as the medians of all possible appropriate combinations of the data and are associated with many non-parametric test procedures. In other words, $\hat{\Delta}$ are the medians of all possible pairwise differences between x values and y values. There are $n \cdot m$ such pairwise differences.

$$\hat{\Delta} = median\left[y_i - x_j\right] \quad (35)$$

where for x_i: $i = 1,\ldots n$ and for y_j: $j=1,\ldots m$.

The estimator $\hat{\Delta}$ is a median-unbiased estimator of the difference in medians, which means that the probability of underestimating or overestimating the difference between the median of x and the median of y is exactly one-half. It is also a resistant estimator.

To find the confidence interval on $\hat{\Delta}$ for a rank-sum test for small sample sizes, a table of quantiles (p-values) for the rank-sum test statistic W_{rs} (for instance, Table B4, Appendix B in Helsel and Hirsch, 1992) is entered to find the critical value x^* having a p-value nearest to $\alpha/2$. This critical value is then used to compute the ranks R_u and R_l corresponding to the pairwise differences at the upper and lower confidence limits for $\hat{\Delta}$. These limits are the R_lth ranked data points going from either end of the sorted list of $N = n \cdot m$ pairwise differences.

$$R_l = x^* - \frac{n \cdot (n+1)}{2} \quad (36)$$

$$R_u = N - R_l + 1 \;\; for \;\; N = n \cdot m \quad (37)$$

When large-sample approximation to the rank-sum test is used, a critical value $z_{\alpha/2}$ from the table of standard normal quantiles determines the upper and lower ranks of the pairwise differences corresponding to the ends of the confidence interval. Those ranks are:

$$R_l = \frac{N - z_{\alpha/2} \cdot \sqrt{\frac{N(n+m+1)}{3}}}{2} \quad (38)$$

$$R_u \begin{cases} = \dfrac{N + z_{\alpha/2} \cdot \sqrt{\dfrac{N(b+m+1)}{3}}}{2} + 1 \\ = N - R_l + 1 \end{cases} \quad (39)$$

1.4.2.2 The Hodges-Lehmann estimator $\hat{\Delta}$ for paired groups

Here, $\hat{\Delta}$ is the median of the $n(n + 1)/2$ possible pairwise averages:

$$\hat{\Delta} = \text{median } [A_{ij}] \quad (40)$$

where $A_{ij} = [\, D_i + D_j \,]$

When the differences are symmetric, $\hat{\Delta}$ associated with the signed-rank test more efficiently measures the additive difference between two data groups than does the sample median of the differences D_{med}.

For a small sample size, a table of quantiles (p-values) for the signed-rank statistic W^+ (e.g., Table B6, Appendix B in Helsel and Hirsch, 1992) is examined to find the critical value x' having p-value nearest to $\alpha/2$. The critical value is then used to compute the ranks R_u and R_1 corresponding to the pairwise averages A_{ij} at the upper and lower confidence limits for $\hat{\Delta}$. These limits are the R_1th ranked A_{ij} going from either end of the sorted list of $n\ (n + 1)/2$ differences.

$$R_1 = x' \quad \text{for } x' = (\alpha/2)\text{th quantile of signed-rank test statistic} \tag{41}$$

$$R_u = x + 1 \quad \text{for } x = (1 - \alpha/2)\text{th quantile of signed-rank test statistic} \tag{42}$$

For the large-sample approximation, a critical value $z_{\alpha/2}$ obtained from the table of standard normal quantiles determines the upper and lower ranks of the pairwise averages A_{ij} corresponding to the ends of the confidence interval. Those ranks are:

$$R_1 = \frac{N - z_{\alpha/2} \cdot \sqrt{\frac{n(n+1)(2n+1)}{6}}}{2} \tag{43}$$

$$R_u = \frac{N + z_{\alpha/2} \cdot \sqrt{\frac{n(n+1)(2n+1)}{6}}}{2} + 1 = N - R_1 + 1 \tag{44}$$

where $N = n\ (n + 1)/2$.

1.4.2.3 The estimator for the sign test

For the sign test, the median difference is the most appropriate measure of how far from 'equality' the two groups are in their original units. Half of the differences are larger and half smaller than the median. A confidence interval on this difference is simply the confidence interval on the median.

1.4.3 Detecting and estimating trends

When an inspection of a time series represented graphically indicates a simpler linear increase or decrease over time, the trend may be analysed with a linear regression of the variables against time followed by a t-test to test if the true slope is zero. However, a lack of normality in distribution, the seasonality effect, and serial

correlation in the time series are constraints on the application of the t-test. In this case, the recommended non-parametric tests are:
(a) a Mann-Kendall test for trend when no seasonal effect is present;
(b) a seasonal Kendall test for trend when seasonality is present and seasons are homogeneous with respect to the trend direction; or
(c) a Mann-Kendall test applied for each season when seasons are not homogeneous with respect to trend direction.

Mann-Kendall and seasonal Kendall tests for trend are independent of the type of distribution. Only the relative magnitudes of the data in the time series rather than their measured values are used. Hence, these tests allow incorporation of missing data values, data reported as a trace, or less than the method detection limit, by assigning them a common value that is smaller than the smallest measured value in the data set.

1.4.3.1 The Mann-Kendall test

The Mann-Kendall test described below is suitable for one datum per time period, where a time period may be a day, week, month, or four quarter of the year, such as the case study presented in **Paper V**. For multiple data values per time period, see Gilbert (1987).

The Mann-Kendall test can be viewed as a non-parametric test for zero slope of the linear regression of time-ordered data versus time (Gilbert, 1987).

Depending on the sample size n ($n \le 40$ or $n > 40$), two alternative procedures, given in Gilbert (1987), can be used.

Number of data values ≤ 40

Consider the data in the order in which they were collected over time as: $x_1, x_2, \ldots, x_n$, where x_i is the datum at time i. Then, determine the sign of all $n\ (n-1)/2$ possible differences $x_j - x_k$, where $j > k$. These differences are $x_2 - x_1, x_3 - x_1, \ldots, x_n - x_1, x_3 - x_2, x_4 - x_2, \ldots, x_n - x_2, x_n - x_{n-1}$.

If sgn $(x_j - x_k)$ is an indicator function that takes on the values 1, 0, or −1 according to the sign of $x_j - x_k$:

$$\mathrm{sgn}(x_j - x_k) = \begin{cases} 1 \ \mathit{if} \ x_i - x_k > 0 \\ 0 \ \mathit{if} \ x_i - x_k = 0 \\ -1 \ \mathit{if} \ x_i - x_k < 0 \end{cases} \tag{45}$$

To compute the Mann-Kendall S statistic:

$$S = \sum_{k=1}^{n-1} \sum_{j=k+1}^{n} {}' \operatorname{sgn}(x_j - x_k) \tag{46}$$

The Mann-Kendall S statistic is the number of positive differences minus the number of negative differences.

The null hypothesis H_0 of no trend is rejected in favour of one of the hypotheses as follows:

H_1: upward trend Reject H_0 if $S > 0$ and the probability $|S| < \alpha$
H_2: downward trend Reject H_0 if $S < 0$ and the probability $|S| < \alpha$
H_3: upward/downward trend (two-tailed test) Reject H_0 if (2 · probability $|S|$) $< \alpha$

A table for probabilities for the Mann-Kendall test for trend is used (e.g., Table A18, Appendix A in Gilbert, 1987).

<u>The large-sample approximation for Mann-Kendall test</u>

This approximation is used when $n > 40$ or when $n \geq 10$ unless there are many tied data values. Compute S using Eq. 20. Compute the variance of S by the following equation, which takes into account that ties may be present:

$$VAR(S) = \frac{1}{18}\left[n(n-1)(2n+5) - \sum_{p=1}^{g} t_p (t_p - 1)(2t_p + 5) \right] \tag{47}$$

where:

g = number of tied groups
t_p = number of data in the pth group

S and $VAR(S)$ are used to compute the test statistic z as follows:

$$Z = \begin{cases} \dfrac{S-1}{[VAR(S)]^{1/2}} & if \quad S > 0 \\ 0 \quad if \quad S = 0 & \\ \dfrac{S+1}{[VAR(S)]^{1/2}} & if \quad S < 0 \end{cases} \tag{48}$$

1.4.3.2 Sen's non-parametric estimator of slope and confidence interval

Sen's method (Sen, 1968b in Gilbert, 1987) is not greatly affected by gross errors or outliers, and it can be computed when data are missing.

Sen's estimator is closely related to the Mann-Kendall test (Gilbert, 1987).
Compute the N' slope estimates, Q:

$$Q = \frac{x_{i'} - x_i}{i' - i} \tag{49}$$

where $x_{i'}$ and x_i are data values at times i' and i, respectively, and where $i' > i$.

The median of these n' values of Q is Sen's estimator of slope. If there is only one datum in each time period, then $n' = n\ (n - 1)/2$, where n is the number of time periods (for more than one datum in each time period, see Gilbert, 1987). If an x_i is below the detection limit, one half the detection limit may be used for x_i.

A 100(1–α)% two-sided confidence interval about the true slope may be obtained by a non-parametric technique also given by Sen (1968b) or by a simpler procedure based on the normal distribution, valid for n as small as 10 unless there are many ties (Gilbert, 1987). This method is as follows.

(a) Choose the confidence coefficient α and find $Z_{1-\alpha/2}$ in a table for cumulative normal distribution (for instance Table A1, Appendix in Gilbert, 1987).
(b) Compute the following.
(c) The lower and upper limits of the confidence interval are M_1th largest and $(M_2 + 1)th$ largest of the n' ordered slope estimates, respectively.

$$C_\alpha = Z_{1-\alpha/2}\left[VAR(S)\right]^{1/2} \tag{50}$$

where $VAR(S)$ is computed from Eq. 4.21.

1.4.3.3 The seasonal Kendall test

If seasonal cycles are present in the data sets, tests for trend that remove these seasonal cycles, or at least are not affected by them, are preferable. The non-parametric seasonal Kendall test was developed by Hirsch et al. (1982), and further discussed by several authors (Smith et al. 1982; van Belle & Hughes, 1984; Gilbert, 1987). This test may be used even though there are values which are missing, tied, or below the analytical detection limit.

The seasonal Kendall test is a multivariate extension of the Mann-Kendall test in which each season is treated as a separate and independent variable.

The authors of the seasonal Kendall test (Hirsch et al., 1982) suggested at least three years of data, where 12 seasons are available, are necessary to apply the test. Conceptually, the seasonal Kendall test may also be used for other 'seasons' such as quarterly sampling per year (Gilbert, 1987). However, the required degree of approximation to a standard normal distribution table has not, apparently, been discussed in the literature. Hirsch et al. (1982) presented a technique that can be used to obtain the exact distribution of the seasonal Kendall test statistic for any combination of seasons and years, where an exact test is important.

For each season *i*, data collected over years are used to compute the Mann-Kendall statistic *S*.

$$S_i = \sum_{k=1}^{n_i-1} \sum_{l=k+1}^{n_i} \operatorname{sgn}(x_{il} - x_{ik}) \tag{51}$$

where $l > k$ and n_i is the number of data (over years) for season *i*.

$$\operatorname{sgn}(x_{il} - x_{ik}) = \begin{cases} 1 \ \textit{if} \ x_{il} - x_{ik} > 0 \\ 0 \ \textit{if} \ x_{il} - x_{ik} = 0 \\ -1 \ \textit{if} \ x_{il} - x_{ik} < 0 \end{cases} \tag{52}$$

The simplified equation for calculating $VAR(S_i)$ when only one datum is available in each specific time period in season i is computed as follows:

$$VAR(S_i) = \frac{1}{18}\left[n_i(n_i - 1)(2n_i + 5) - \sum_{p=1}^{gi} t_{ip}(t_{ip} - 1)(2t_{ip} + 5) \right] \tag{53}$$

where g_i is the number of groups of tied (equal-valued) data in season *i*, and t_{ip} is the number of tied data in the *p*th group for season *i*. After S_i and $VAR(S_i)$ are computed, the *K* seasons are pooled across:

$$S' = \sum_{i=1}^{K} S_i \tag{54}$$

and the *Z* value is computed as:

$$Z = \begin{cases} \dfrac{(S'-1)}{[VAR(S')]^{1/2}} \ \textit{if} \ S' > 0 \\ 0 \ \textit{if} \ S' = 0 \\ \dfrac{(S'+1)}{[VAR(S')]^{1/2}} \ \textit{if} \ S' < 0 \end{cases} \tag{55}$$

To test H_0 against the H_1 of either an upward or downward trend (two–tailed test), we reject H_0 if the absolute value of *Z* is greater than $Z_{1-\alpha/2}$, where $Z_{1-\alpha/2}$ is from a standard normal table for cumulative normal distribution (values of *p* corresponding to Z_p for the normal curve). If the alternative hypothesis is for an upward trend at the α level (one-tailed test), we reject H_0 if Z is positive and the absolute value of Z is greater than $Z_{1-\alpha}$. If the alternative hypothesis is for an downward trend at the α level (one-tailed test), we reject H_0 if Z is negative and the absolute value of Z is greater than $Z_{1-\alpha}$.

1.4.3.4 The seasonal Kendall slope estimator

To estimate the magnitude of a trend, the non-parametric seasonal Kendall slope estimator is appropriate. The individual slope estimates N_i are computed for the ith season:

$$Q_i = \frac{x_{il} - x_{ik}}{l - k} \tag{56}$$

where x_{il} is the datum for the ith season of the lth year, and x_{ik} is the datum for the ith season of the kth year, and $l > k$.

Calculate Q_i for each of the K seasons. Then rank the $N'_1 + N'_2 + \ldots\ldots + N'_k = N'$ individual slope estimates and find their median (the seasonal Kendall slope estimator).

A $100(1-\alpha)\%$ confidence interval about the true slope is obtained in the same manner as for the Sen's estimator of slope.

1.4.3.5 Homogeneity of trends in different seasons

The reason for applying a test for homogeneity of trend direction in different seasons (Van Belle and Hughes, 1984) is that, if the trend is upward in one season and downward in another, the seasonal Kendall test and slope estimator will be misleading. In this case, trend tests should be conducted separately by season. The procedure for obtaining the homogeneous chi-squared (X^2_{hom}) is:

$$X^2_{\mathrm{hom}} = X^2_{total} - X^2_{trend} = \sum_{i=1}^{K} Z_i^2 - K\bar{Z}^2 \tag{57}$$

where

$$Z_i = \frac{S_i}{[VAR(S_i)]^{1/2}} \tag{58}$$

S_i is the Mann-Kendall statistic, computed using data collected over some years, during the ith season, and

$$\bar{Z} = \frac{1}{K}\sum_{i=1}^{K} Z_i \tag{59}$$

If X^2_{hom} exceeds the critical α value for the chi-square distribution with K–1 degrees of freedom, we reject the null hypothesis H_0 of homogeneous seasonal trends over time (trends in the same direction and of the same magnitude). In this case, it is best to compute the Mann-Kendall test and Sen's slope estimator for each season. Otherwise, the X^2_{trend} is referred to the chi-square distribution with 1 degree of freedom to test for a common trend in all seasons.

1.4.4 Winters' method for time series modelling and forecasting

Forecasting and smoothing methods for time series analysis are based upon whether the patterns in the time series are static (constant over time) or dynamic (change over time), the nature of the trends and seasonal components, and how far ahead one wishes to forecast.

Simple forecasting and smoothing methods model components, which are easily identified in a time series plot of the data. ARIMA (Auto-Regressive Integrated Moving Average) modelling also makes use of patterns in the data, but these patterns may not be readily apparent in a plot of the data. Instead, ARIMA modelling uses differencing, autocorrelative, and partially autocorrelative functions to help identify an acceptable model.

Winters' Method (Minitab® 1997) smoothes data using Holt-Winters' exponential smoothing and provides short to medium range forecasting. It is suitable for time series where trend and seasonality are present, with these two components being either additive or multiplicative.

Winters' Method calculates dynamic estimates for three components: level, trend, and seasonal, for each period, to generate forecasts. It uses three weighting, or smoothing, parameters to update the components at each period. Initial values for the level and trend components are obtained from a linear regression with time. Initial values for the seasonal component are obtained from a dummy-variable regression using de-trended data (Minitab® 1997).

The Winters' Method smoothing equations are as follows.

Additive model:

$$L_t = \alpha(Y_t - S_{t-p}) + (1-\alpha)[L_{t-1} + T_{t-1}] \quad (60)$$

$$T_t = \gamma(L_t - L_{t-1}) + (1-\gamma)T_{t-1} \quad (61)$$

$$S_t = \delta(Y_t - L_t) + (1-\delta)S_{t-p} \quad (62)$$

$$\hat{Y}_t = L_{t-1} + T_{t-1} + S_{t-p} \quad (63)$$

Multiplicative model:

$$L_t = \alpha\left(Y_t / S_{t-p}\right) + (1-\alpha)\left[L_{t-1} + T_{t-1}\right] \tag{64}$$

$$T_t = \gamma\left(L_t - L_{t-1}\right) + (1-\gamma)T_{t-1} \tag{65}$$

$$S_t = \delta\left(Y_t / L_t\right) + (1-\delta)S_{t-p} \tag{66}$$

$$\hat{Y}_t = (L_{t-1} + T_{t-1}) / S_{t-p} \tag{67}$$

where:

L_t is the level at time t, α is the weight for the level, T_t is the trend at time t, γ is the weight for the trend, S_t is the seasonal component at time t, δ is the weight for the seasonal component, p is the seasonal period, Y_t is the data value at time t, and $\hat{Y}_t$ is the fitted value, or one-period-ahead forecast, at time t.

Measures of accuracy of forecasting with Winters' Method include the Mean Absolute Percentage Error (MAPE), Mean Absolute Deviation (MAD), and Mean Square Deviation (MSD).

$$MAPE = \frac{\sum\left|\left(y_t - \hat{y}_t\right)/y_t\right|}{n} x100 \quad (y_t \neq 0) \tag{68}$$

$$MAD = \frac{\sum_{t=1}^{n}\left|y_t - \hat{y}_t\right|}{n} \tag{69}$$

$$MSD = \frac{\sum_{t=1}^{n}\left(y_t - \hat{y}\right)^2}{n} \tag{70}$$

where y_t equals the actual value, $\hat{y}_t$ equals the forecast value, and n equals the number of forecasts.

MAPE measures the accuracy of titted time series values. It express accuracy as percentage.

MAD measures the accuracy of fitted time series values. It expresses accuracy in the same unitis as the data, which helps conceptualize the amount of error.

MSD is very similar to a mean squared error (*MSE*), a commonly-used measure of accuracy of fitted time series values. The advantage of *MSD* over *MSE* is that *MSD* is always computed using the same denominator, *n*, regardless of the model, which makes it possible to compare *MDS* values across models.

For all three measures (*MAPE*, *MAD* and *MSD*), the smaller the value, the better the fit of the model.

2 SITE DESCRIPTION

2.1 Spillepeng landfill

The description of the Spillepeng landfill presented below is based on the Spillepeng landfill annual environmental reports (SYSAV, 1991; 1992; 1993; 1994; 1995; 1996; 1997; 1998 and 1999).

Site geology and hydrogeology described here were based on analysis and interpretation of the geological and hydrogeological field data generated during different periods of landfill planning and construction (VBB VIAK, 1976; 1978; 1984; 1985; 1988), and Swedish geological survey reports of southern Sweden (Ringberg, 1975; 1980; Gustafsson, 1993; SGU 1998; Sivhed et al., 1999).

2.1.1 The old and the new landfills: Main features

The Spillepeng landfill is located at Lomma Bay in the Sound (Öresund), which separates southern Sweden from Denmark. The site comprises an old landfill, which was located on the original coast, and a new landfill, located on land in Lomma Bay itself (Figs. 2.1 and 2.2). The land for construction of the new landfill was reclaimed from the Bay using a seawall to allow engineering of the base of the new landfill (Fig. 2.3).

The old landfill occupies an area of 65 ha and was constructed on marshland. The site is surrounded by surface water on three sides: the Sege River to the south, Lomma Bay to the west and the Kalina River to the north.

A layer of artificial fill underlies this landfill. Although good records of the fill material were not maintained at the time, it is understood that during the 1950s, the site was used for disposal of excavated soil, municipal solid waste (MSW) and industrial waste forming a layer that varies in thickness from 2 to 10 m.

During the 1970s, construction of a sanitary landfill at the site was planned, and site hydrogeological investigations were undertaken in 1976 (VBB VIAK, 1976). The site was then prepared for operation as a MSW landfill and operated as such from 1977. The old landfill received MSW, industrial waste and incineration residues (bottom ash and fly ash).

The old landfill was closed in 1989, after which it was capped with a one-metre thickness of glacial till and vegetation was planted on top. The average height of the old landfill is 12 metres above sea level (masl) and is 20 masl at the highest point (Fig. 2.3).

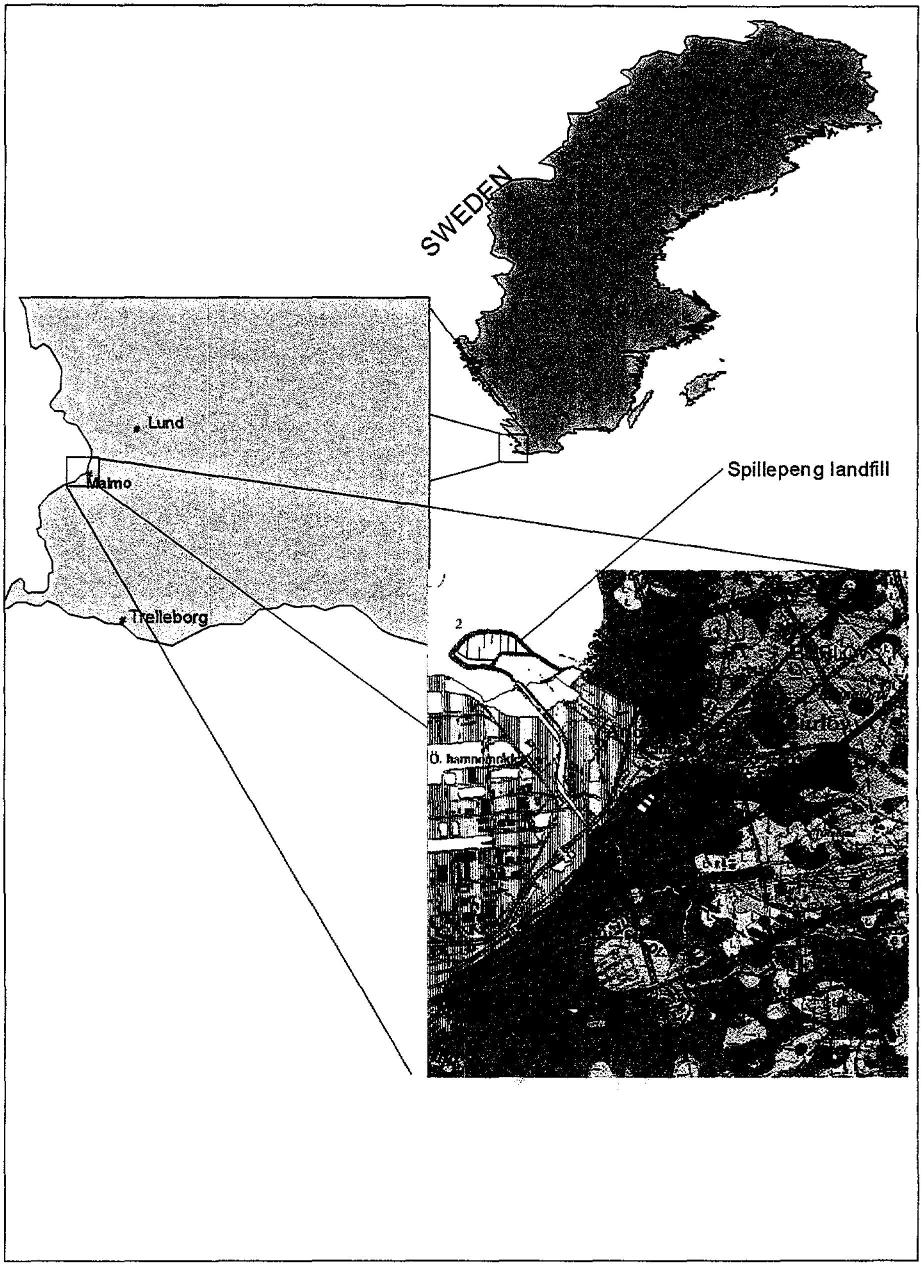


Figure 2.1 Location of Spillepeng waste management park: Southern Sweden, Lomma Bay.

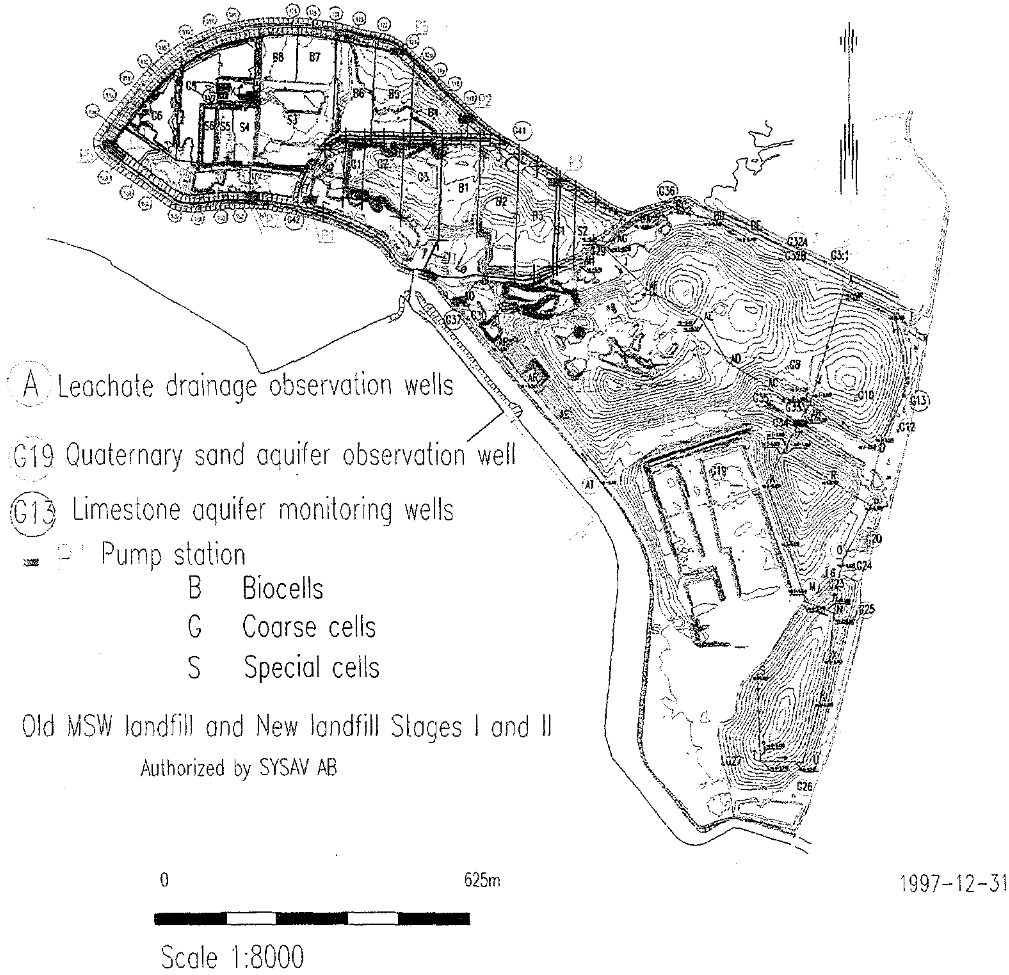
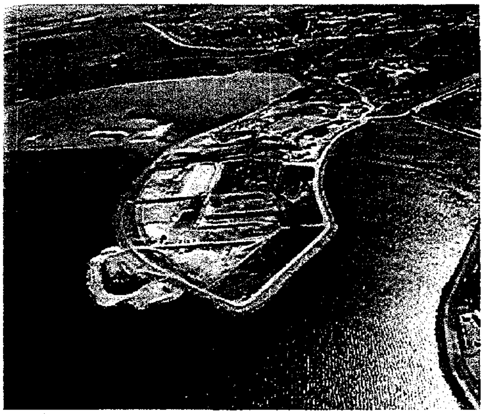


Figure 2.2 Topographic map of Spillepeng site, showing the waste filling in Stage II (1997). Observation wells are those used only for water head monitoring, with no water sampling.

No liner was installed in the old landfill to prevent migration of leachate into the underlying groundwater system. In order to establish a form of groundwater pollution control, a leachate and groundwater drainage system was installed in the sandy material at the base of the landfill (Fig. 2.4). Since the drainage system was located below the water table, groundwater is collected in these drains in addition to leachate. It has been estimated that 0.5–0.8 L s^{-1} of groundwater enters this drainage system (SYSAV 1991).

a



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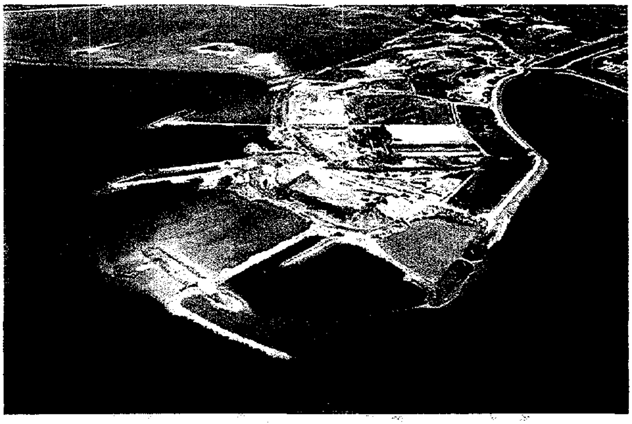

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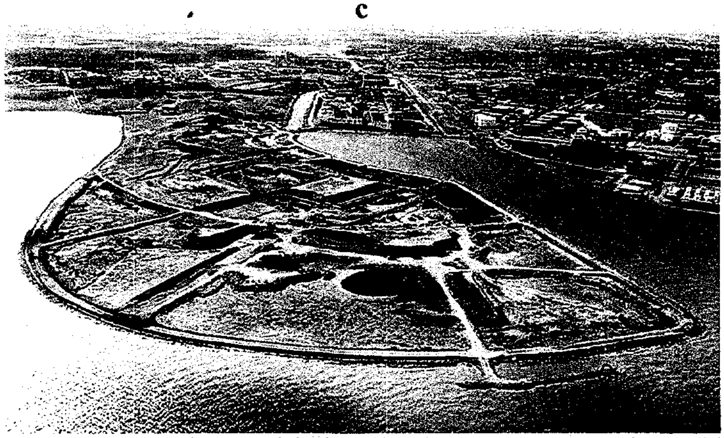


Figure 2.3 Spillepeng expansion. Aerial view showing Stages I, II and III with different phases of the construction of Stage III. a) Beginning of construction (1998). b) Embankment construction (1998). c) Bottom of future cells and sorting/storage areas (1999).

In order to reduce the amount of groundwater discharging into the old landfill, a number of corrective engineering works were undertaken during drainage construction:

- Groundwater flow into the landfill was restricted from the south by the Sege River embankment (Fig. 2.4), which runs along the riverbank adjacent to the old landfill.
- Groundwater flow into the landfill was restricted from the west by a wall constructed between the old landfill and the new landfill.
- The sugar factory SSA drainage ditch has been removed and the Kalina River relocated to flow past the northern side of the site.

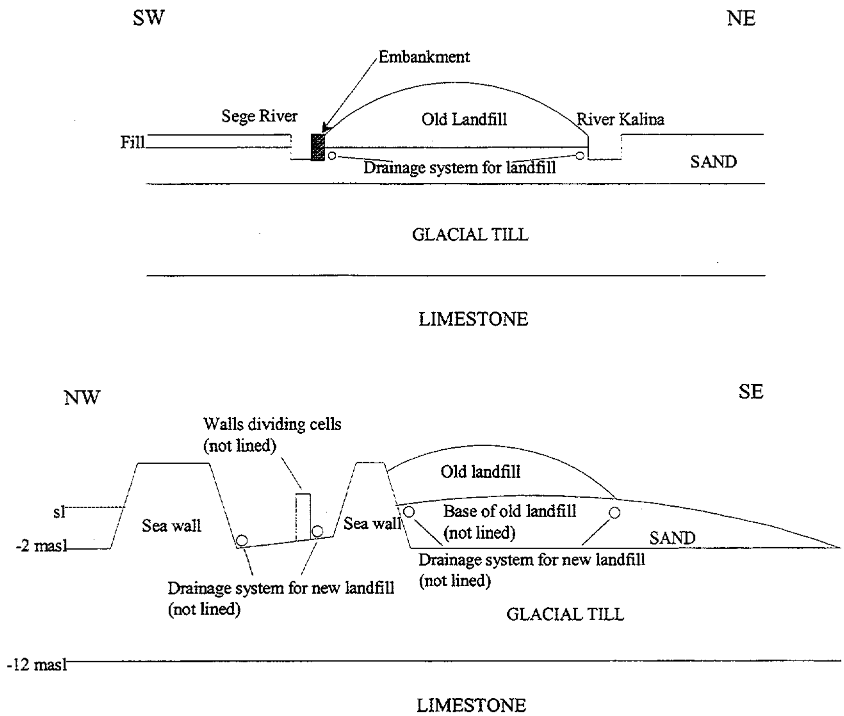


Figure 2.4 Schematic cross-section of Spillepeng landfill.

The eastern end of the old course of the Kalina River was landscaped. As part of the landscaping processes after closing the old landfill in 1989, a HDPE liner was laid in this area.

2.1.2 New Landfill: Stages I, II and III

After closure of the old landfill, the site was subsequently extended into Lomma Bay. This new landfill has a total area of 56 ha and was constructed in three stages: Stage I

(16 ha), Stage II (22 ha) and Stage III (18 ha). Each stage included landfill cells and areas for sorting, processing and storage of recovered materials.

As part of the engineering works used to construct the new landfill, the top layer of the underlying sand was removed and hence the base of the new landfill lies on glacial till (Fig. 2.4). Consequently, the bases of the cells in the new landfill had an elevation of -1.5 masl. This base elevation was designed to produce a 'hydraulic trap' to prevent landfill leachate from migrating into Lomma Bay. As with the old landfill, no liner was emplaced at the base of the landfill cells.

A wall was constructed around the perimeter of each stage of the new landfill (i.e., Stages I, II and III) to isolate the landfill from Lomma Bay and from a part of the old landfill where it is 10 m high (8.5 masl). The wall was constructed using construction and demolition waste plus sandy material excavated from the new landfill site. It was covered with a compacted clay layer that had been engineered to achieve a maximum hydraulic conductivity of 10^{-7} m s^{-1}, a plastic liner and a gravel layer. Stone blocks weighing approximately 600 kg each were placed on the seaward side of the wall to protect it from erosion by the sea.

Within the new expanded landfill, cells were created that are separated from each other by unlined walls. The cells are grouped according to the type of waste matter. Each group of cells forms a discrete drainage unit connected to a dewatering sump. Similarly to the old landfill, the leachate is pumped from the sumps and sent to the municipal wastewater treatment plant.

2.1.3 Drainage system

The Spillepeng landfill drainage system is shown in Fig. 2.5. The drainage system directs the water into collection sumps. At the collection sumps, the water is pumped to the Sjölunda wastewater treatment plant located less than 500 m from the site.

Pump stations P4 and P4b serviced the old landfill drainage system. Each group of cells and each stage of the new landfill was serviced by an independent collection sump and pump station system. The pump stations located in, and servicing Stage I were: station P1 for coarse waste cells G1–G3 and the sorting-storage area contiguous to the coarse waste cells; station P2 for biocells B1–B3 and station P3 for special waste cells S1 and S2.

At Stage II the pump stations were: P5 for the sorting-storage area; P6 for biocells (B4–B7) and P7 for special waste cells (S3–S5).

Over recent years, leachate drained from special waste cells in the Stage II area has been diverted for on-site treatment, due to the high content of heavy metals, particularly lead, in the leachate.

A number of observation wells where the water head is monitored monthly are situated along the drainage system of the old landfill (35 locations) and of the new landfill (17 locations).

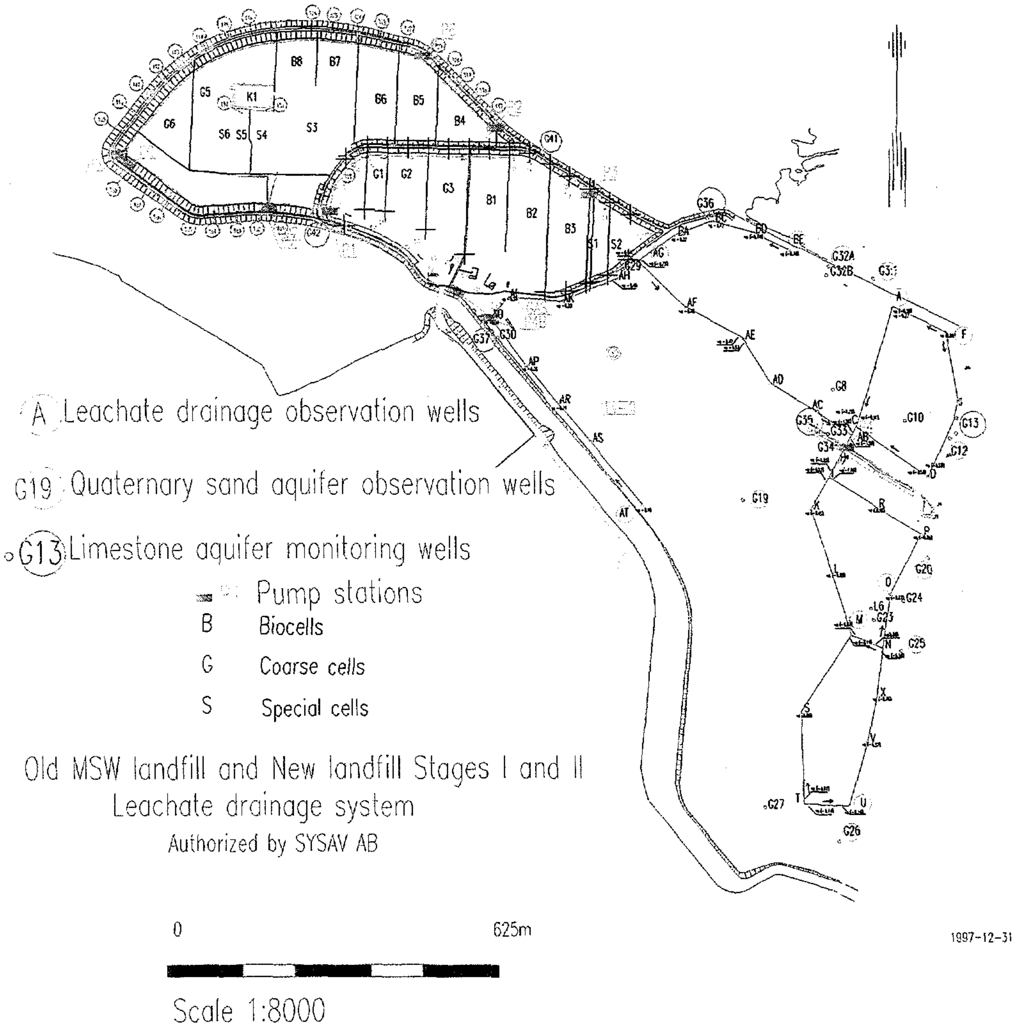


Figure 2.5 Spillepeng landfill drainage system and cell division in Stages I and II.

Since both the old and new landfills are unlined, protection of water bodies from eventual pollution by leachate relies on maintenance of a hydraulic gradient toward the landfill.

In such a 'hydraulic trap' design, good maintenance of the drainage systems is of paramount importance and, from a long term perspective, can be seen as the most important aspect to be considered.

2.1.4 Waste disposal and recovery

Wastes arriving at the site (1990–1998) are shown in Tables 2.1 and 2.2 and Fig. 2.6.

Table 2.1 Total waste received by Spillepeng annually (x 1000 tonnes).

	1990	1991	1992	1993	1994	1995	1996	1997	1998
Received*	511.1	668.5	377.5	359.8	194.5	217.8	273.4	466.2	634.0
Landfilled	205.6	216.0	182.0	154.0	119.0	105.0	124.0	124.1	138.7
Recycled*	305.5	452.5	195.5	205.8	75.5	112.8	149.4	342.1	495.3

*Including excavated soil.

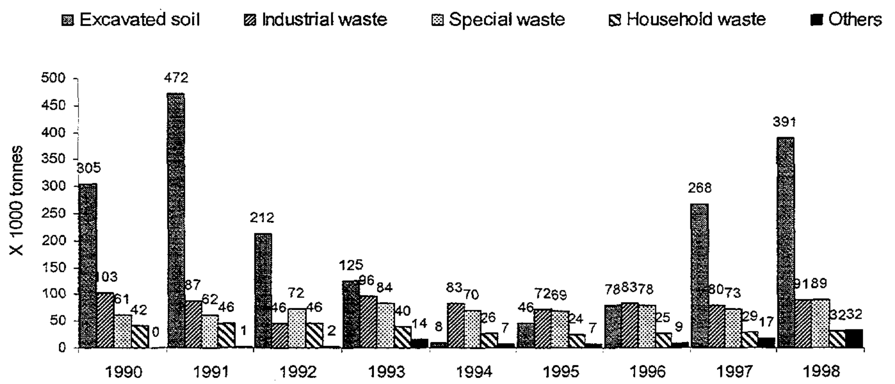


Figure 2.6 Waste streams/materials arriving at Spillepeng. High amounts of excavated soils during the first years and during the last years are related to the construction of Stage II and III respectively (SYSAV, 1990; 1991; 1992; 1993; 1994; 1995; 1996; 1997; 1998).

Table 2.2 Waste (x 1000 tonnes) classified by SFS (1998), mostly destined to special cells in Spillepeng landfill. During 1990-1992, only total values are available.

Waste (code n° in SFS, 1998)		1990	1991	1992	1993	1994	1995	1996	1997	1998
Asbestos/cement sheets	(17 01 05)				1.73	1.05	1.21	1.32	1.15	1.53
Clinical waste	(18 00 00)				0.17	0.12	0.15	0.08	0.07	0.05
Polluted soil	(17 05 00)				0.85	0.46	0.70	0.53	0.27	0.12
Blasting grit	(12 02 01)				0.08	0.13	0.12	0.10	0.16	-
Restaurant Sludge[a]	(02 00 00)				11.43	12.17	12.45	12.51	12.06	12.69
Others					16.94	6.54	6.78	17.19	18.26	26.42
	SUB TOTAL	13.65	14.8	13.26	31.2	20.47	21.41	31.73	31.97	40.81
Bottom ash	(19 01 01)									
Fly-ash	(19 01 03)	*47.66**	*47.66**	*47.66**	53.2	49.9	47.23	46.44	41.09	48.11
	TOTAL	61.31	62.46	60.92	84.4	70.37	68.64	78.17	73.06	88.92
Recycled bottom ash		27.52	23.45	29.61	38.80	36.90	33.83	34.74	29.62	29.61

[a] Co-disposed with MSW in biocells; * The period 1990-1992 was estimated.

Special cells received mostly fly ash and the residue fraction of bottom ash from MSW incineration (mass-burn type incinerator). Other types of waste deposited in these cells were contaminated soils, blasting grit, asbestos-cement sheets, and waste generated at hospitals and medical centres. Special cells filled in Stage I occupy 1.5 ha and were closed in December 1993. The bulk density of the freshly emplaced waste in these cells was approximately 0.69 (weight/volume). After completion, these cells were covered with 1 m of glacial till material. In Stage II, special cells occupy about 1.7 ha and are still in operation.

Biocells have received MSW, some industrial wastes and sludges from the food industry and restaurants. In Stage I, biocells occupy 5 ha and were closed in April 1994. The bulk density of the freshly emplaced waste in these cells was approximately 0.88 (weight/volume). Upon completion of filling, 75% of these cells were covered with 0.8 m of glacial till. The remaining 25% was left uncovered to allow for infiltration of sludges. Biocells in Stage II occupy 3.5 ha and are still in operation. Biogas is extracted from the biocells and used for heating purposes.
Coarse waste cells located in Stage I received mostly inert/coarse waste (demolition and construction waste). These cells occupy 3.5 ha and were closed in April 1993. Upon completion, these cells were covered with 0.8 m of glacial till material.

Due to an increase over time in the degree of sorting and recovery of several materials, the area originally reserved for coarse waste, cells G4–G6 in Stage II (5 ha), has been used instead for storage, sorting and treatment of wastes that are not landfilled.

2.1.5 Site geology

The 1:50,000 geology map (Fig. 2.7) shows that coastal areas of Malmö are largely underlain by artificial fill. This area includes the Spillepeng landfill site and a larger area to the west of the Sege River. It would appear that this fill was generally emplaced over low-lying marshy areas underlain by sand and gravel deposits.

The fill materials at the old Spillepeng landfill site were deposited directly onto deposits consisting of sands, clays and peat. These deposits overlie a relatively thick glacial till beneath which a Danian Limestone is found (Table 2.3).

The glacial till is between 10- and 14-m thick. The geological description of the till indicated that the sandy intermorainal deposits described in the Swedish geological survey memoirs (SGU, 1998) are absent at this location. The till was typically described as a silty, sandy clay or, less frequently, as a silty clay. Fragments of chalk and pieces of flint were recorded in places. No consistent vertical or horizontal variation in lithology was apparent, although a few descriptions indicated that in some places the sandy clay occurred at the top of the till. The limestone in the area was noted as being grey-white, hard, coarse-grained with a high silica content, containing flint horizons. Fracture frequency was observed to be highest towards the surface of this bedrock. On the eastern side of the landfill, limestone bedrock was found at an elevation of approximately -15 masl, rising to -12 masl at the western end of the new extension. West of Spillepeng, the bedrock rises to between -10 and -5 masl in the Malmö harbour area (VBB VIAK, 1976; 1978 and; Sevhed, 1999).

Based on particle size distribution analyses (VBB VIAK, 1984) it was found that only four out of 11 samples contained any clay-sized particles and the clay content in those samples was less than 18 %. The majority of the samples were dominated by silt-sized particles, although up to 60 % sand was also present in some samples. Nine out of the 11 samples contained less than 7 % of gravel-sized material

Table 2.3 Spillepeng landfill: Site geology and hydrogeological properties.

Age	Name	Thickness (m)	Lithology	Hydrogeological Properties
1950s	Artificial Fill	0 – >10 m	Glacial till, silty sand, waste material	Variable
Quaternary		1.7 to >11.7 m	Sand and gravel, silt, peat, gyttja* clay	Variable, aquifer/non-aquifer
		10–14 m	Sandy, silty clay	Non-aquifer
Lower Tertiary	Copenhagen Member Upper Danian		Limestone	Aquifer

* Swedish word for a nutrient rich sedimentary peat, consisting mainly of plankton and other plants, animal residues and mud.

The elevation at the till surface varies from -1 to -4 masl. Beneath the old landfill site there are a variety of sediments and organic deposits resting on the till. These vary from sands and gravels through to clay, silt, peat and thin gyttja (organic-rich) clay in places. Due to the complex interrelationships of these deposits, they have been grouped together as a single unit, varying in thickness from 1.7 m to 11.7 m.

The sand appears to be the dominant lithic member of this group, whereas the peat, clay and silt deposits may occur above or below the sand and occasionally as thin bands within the sand.

2.1.6 Hydrogeological setting

Four hydrostratigraphic units have been identified at the Spillepeng site. These are:

- Fill and waste material
- Quaternary sand, clay and peat deposits
- Glacial till
- Danian (Late Tertiary) limestone

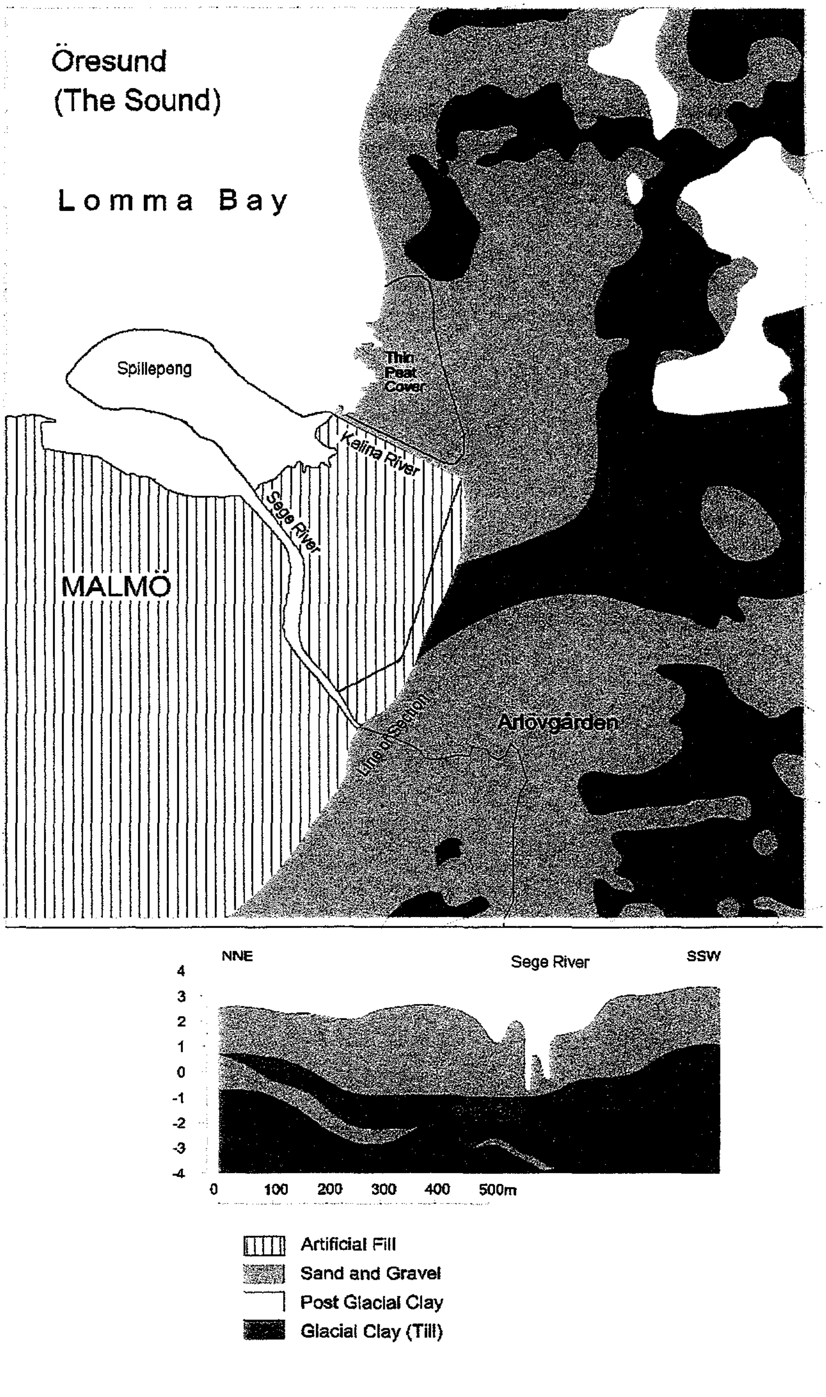


Figure 2.7 Drift geology. Spillepeng site.

All four units are present at the old landfill site, while in the new landfill extension into Lomma Bay, these sand deposits were removed, and emplaced waste and fill materials lie directly on the glacial till. It is noted that waste materials in the old landfill are in direct hydraulic contact with the sand deposits.

Permeameter tests on recovered samples of Quaternary sands typically indicate a saturated hydraulic conductivity (K) of about 4×10^{-4} m s^{-1} (VBB VIAK, 1976).

Additional information on the K of these sands was obtained by analysing the particle size distribution (PSD) curves of three sand samples, using the method selected to estimate K from PSD curves devised by Masch and Denny (1966). The results indicated a range for K of between 2.6×10^{-4} and 9.9×10^{-4} m s^{-1}.

Permeability measurements made on laboratory samples (VBB VIAK, 1976) from the glacial till deposits underlying the sands gave K values between 8.6×10^{-8} and 4.3×10^{-7} m s^{-1}. Further permeability tests (VBB VIAK, 1984) were carried out in till samples taken from four boreholes with different depths. Here, K varied from 2.0×10^{-7} to 2.2×10^{-9} m s^{-1}.

Given the lithological description of this material, it would appear that K values quoted above are consistent with clayey silts and clayey sands. Based on the description of the glacial till, considering the whole extension of the landfill site, the K of the till is likely to fall within the range 0.5×10^{-5} to 2.2×10^{-9} m s^{-1}.

Based on these findings, the limits established by the new EU Directive on the landfill of waste, regarding hydraulic conductivities of the geological barrier required for different classes of landfills (Annex I of the Council Directive 1999/31/EC, 1999) should be considered during expansion of the new landfill in Stage III.

No local data is available for transmissivity (product of hydraulic conductivity and aquifer thickness) of the fractured limestone aquifer. The values quoted for coastal areas in southwest Skåne where Spillepeng is located, lay between 86 and 260 m^2 d^{-1} (Gustafsson, 1993).

2.2 Lidköping District Heating Plant and the Bale Storage Area

The Lidköping district heating plant is located on the shores of Lake Vänern in southwestern Sweden. Lidköping Värmeverk AB operates the incinerator, a baling machine and the area used for storage of waste intended as fuel. The storage area was object of investigations related to the stormwater runoff quality (**Paper IV**).

The plant was designed to use increasing amounts of household and commercial waste produced by the community, as fuel. By burning 50,000 tonnes of waste each year, the plant supplies some 18,000 people in the district with heating.

Two 12-MW bed fluidized boilers (BFB) and two 20-MW oil-fired hot water boilers started operation at this site at the end of 1985. The BFBs were subsequently upgraded to 17 MW each. The upgraded boilers commenced operation in 1993 and 1994, respectively. The BFB boilers are used for the base load throughout the year, whereas the oil-fired boilers are used for peak loads during winter.

Typically, shredded MSW, wood waste, or both are burned 24 hours a day, 5-6 days a week.

The fully automated baling machine manufactures 1,000-kg bales using stretched polyethylene. The size of the bales is 120 x 120 cm. Their weight depends on the type of waste material in them and their moisture content. The compaction achieved during baling results in some leachate draining out of the bales, mainly from those bales containing MSW.

The bale storage area occupies about 1.3 ha close to the incineration plant (Fig. 2.8). At the time when stormwater runoff generated from the bale storage area was studied (**Paper IV**), about 2,500 bales, 90% of them filled with MSW, and 10% filled with used plastic fertilizer sacks, were standing in piles, four bales high.

In all, the piles occupied 1 ha of the asphalt surface, and 0.3 ha of that surface was free of bales.

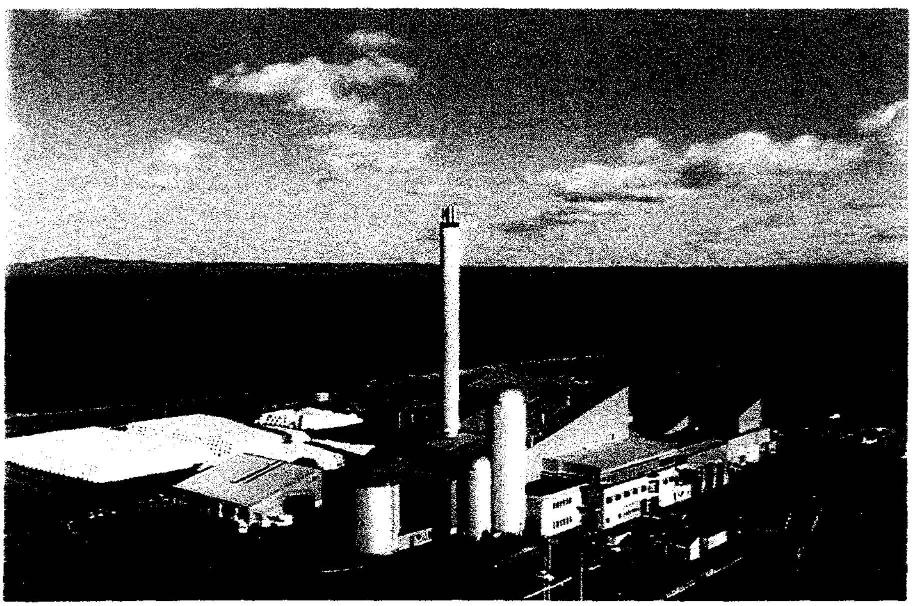


Figure 2.8 Incineration plant in Lidköping, southwestern Sweden. The storage area with bales of waste fuel is shown in the west side of the picture, between the Lake Vänern and the incinerator.

3. PAPER OVERVIEW

Five journal papers are appended to this thesis, addressing the following four main issues:

- the storage of waste fuel, and physical, biological and chemical processes related to risk of spontaneous ignition (**Paper I:** *Physical, Biological and Chemical Processes During Storage and Spontaneous Combustion of Waste Fuel*);

- the consequences of active and capped landfills on the hydrological performance/discharge due to changes in the composition of waste destined for landfilling (**Paper II:** *Hydrological Performance of Fly-ash Co-disposed with Other Special Wastes and MSW Co-disposed with Sludge in Full-scale Cells*);

- stormwater runoff related to different waste handling activities intensified in a modern waste management park (MSW) (**Paper III:** *Stormwater Runoff and Pollutant Transport Related to Activities Carried out in a Modern Waste Management Park* and **Paper IV**: *Stormwater Runoff Pollution at Waste Management Sites Compared to Runoff from Other Land Uses*); and

- statistical procedures appropriate for monitoring of groundwater quality at landfill sites (**Paper V**: *Statistical Analysis of Groundwater Quality Time-Series in a Landfill Site*).

For most non-EU countries, landfill co-disposal remains one of the most common options for treatment of all kinds of sludge. Paper II discusses additionally the fate of sludge water, once it is co-disposed with MSW.

Together, these five papers highlight some of the main constraints faced when planning, implementing and executing environmental control programmes at modern waste management parks where several waste handling practices are carried out, among them the operation of active landfills and the landfill post-closure monitoring programme.

By comparing waste management parks with other urban land uses, such as industrial, traffic and residential, from a stormwater-pollutant transport standpoint (Paper IV), the authors intend to place the waste management parks in a more general context regarding their relative contribution to water pollution.

The objective of the present chapter is to give an overview of the appended papers. Therefore, literature references were excluded. These references are presented and discussed in the original appended papers.

3.1 Storage of Waste as Fuel (Paper I)

The storage in piles of unsorted industrial waste (IND) fuel and refuse-derived fuel (RDF) at two different locations in Sweden was investigated. The objective was to assess the feasibility of waste storage in open areas for later incineration and energy utilization, and to study the physical, biological and chemical processes that occur during storage, by monitoring a number of parameters.

The conclusions were based mainly on two spontaneous combustion events that occurred after six months of storage: one occurring in an IND fuel storage pile and another occurring in a RDF storage pile. An explanation for the spontaneous ignition occurring in the waste piles was developed, based on the observed data and previous studies available in the literature about storage of forest fuel material.

In the storage of IND fuel in piles, it was assumed that there was convective flow in the upper and intermediate levels of the piles, by analogy with the forest fuel studies. The measurements carried out showed higher concentrations of O_2 in the upper and intermediate levels of the pile (Fig. 3.1).

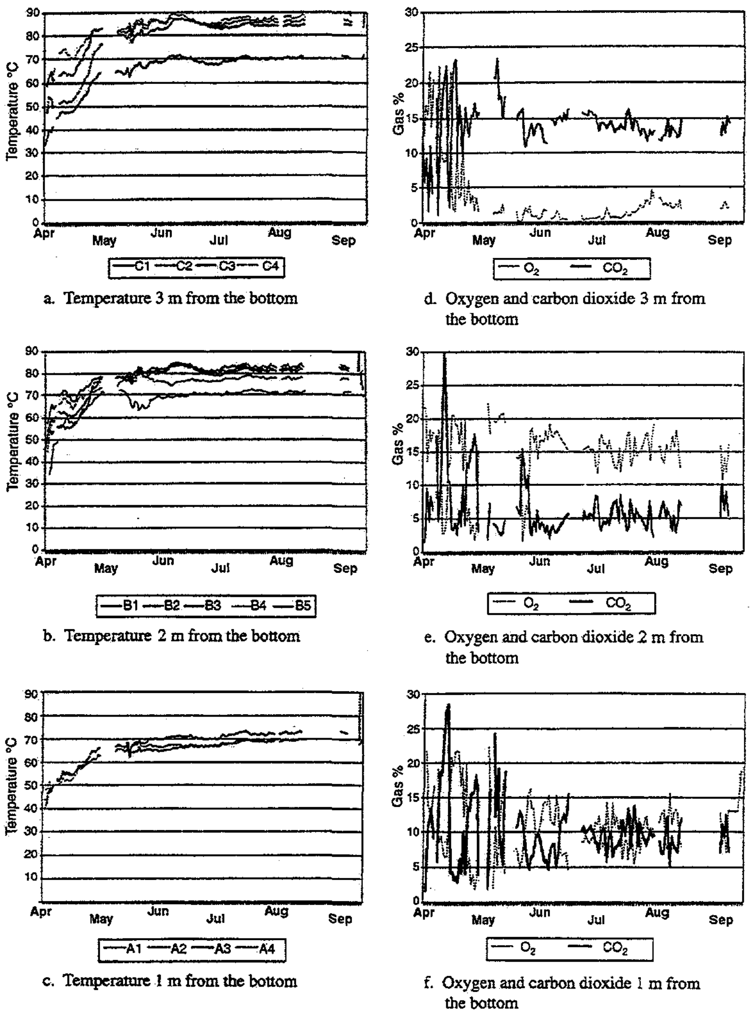
These measurements indicate that the supply of O_2 depends not only on diffusion, but also on other means. The reason the O_2 concentration is higher in the intermediate level of the pile (2 m from the bottom) than in the upper level (3 m from the bottom) is that air inflow occurs through the lateral surface of the pile. The inflow is channelled, and the outflow occurs through the upper surface of the pile.

The higher temperature in the upper, inner part may be due to a flow of heated moisture-bearing gas through this region. Condensation occurs when the gas starts to cool, and heat is delivered to this region by conduction and release of latent heat. In this way, the temperature front will move upwards as moisture is transported towards the upper surface of the pile.

With respect to the degree of compaction achieved when waste piles are constructed, it is concluded that compaction, even when it is made properly, cannot eliminate the risk of self-ignition. Channelling and natural convection create zones with sufficient moisture and oxygen supply to promote biodegradation, leading to heating and spontaneous combustion in both compacted and non-compacted piles.

Considerable changes in temperature and gas composition occur during the first few weeks of storage, reflecting the intense biological activity taking place in the readily-biodegradable fraction of the waste. This raises the temperature into the optimum range for chemical oxidation reactions. Together with the waste's own weight, this promotes settling. Therefore, in a similar manner to landfills, regions with different permeability are expected to develop.

Apparent steady-state conditions established in the pile after the first few months may actually reflect continuous degradation, with increased participation by non-biological chemical processes, further increasing the temperature and reducing the energy content of the waste material. At this point, in the presence of catalytic materials such as metals, optimum conditions for spontaneous combustion are reached.



a. Temperature 3 m from the bottom

d. Oxygen and carbon dioxide 3 m from the bottom

b. Temperature 2 m from the bottom

e. Oxygen and carbon dioxide 2 m from the bottom

c. Temperature 1 m from the bottom

f. Oxygen and carbon dioxide 1 m from the bottom

Figure 3.1 Measurements taken during IND fuel storage. Gas measurements are shown for only one point per level, which is representative of the range observed.

Some practical countermeasures to reduce the risk of fire are to remove coarse materials, such as tyres, and readily degradable waste, such as garden and kitchen waste; avoid metallic materials; ensure piles are lower than 5 m in height; construct the piles parallel to the direction of the prevailing wind; and protect piles from strong winds by providing wind barriers.

While the baling of waste fuels would benefit from further investigation and improvement, it seems to be a more promising strategy than storage in loose piles, in terms of preventing self-combustion, as it reduces the biodegradation rate and, thus, loss of energy values during storage.

3.2 Leachate Generation by Active and Capped Landfills (Paper II)

Since water carries and distributes pollutants in the environment, it is important from both short- and long-term perspectives to understand water-flow mechanisms through landfills. Assessment of the hydraulic characteristics of landfills is a relevant requirement, due to the potential impact on groundwater quality of uncontrolled leachate releases. The need for additional investigations of the hydrological behaviour of landfills of solid wastes other than MSW, particularly special waste comprising the main portion of landfilled waste, is clearly demonstrated by the few references on this matter found in the literature. These are restricted to MSW incineration residues, mainly fly ash. Landfills of special waste (e.g. incineration residues, sheets of asbestos/cement, blasting grit, contaminated soil, clinical waste) are expected to make up the majority of landfill content in the EU countries in the future. Additional knowledge on their hydrological behaviour is required.

In Paper II, the landfill hydrological performance was investigated in active and capped landfill cells of two types:

- special cells where incineration residues from a mass-burn incinerator and other special waste (non-compostable and non-recyclable waste such as asbestos sheeting and contaminated soil) were disposed; and
- biocells where MSW and sludge were co-disposed.

The objectives were:

- to investigate the hydrological performance and the water flow regime in uncovered and covered special cells and biocells, using the HELP model based on a full-scale case (Spillepeng landfill, Stage I);
- to compare special cells and biocells, regarding the water flow regime in the waste domain; and

- to formulate one hypothesis for the fate of the original sludge moisture content following disposal in MSW landfills, using the HELP model and the basic water balance approach.

Special cells

The observed discharge from uncovered special cells, particularly during the first two years, was very high, and accounted for 85% of the precipitation. The ratio between net infiltration and precipitation for the active period was 0.53. Even if all net infiltration became discharge, there was still a water excess of about 32% of precipitation contributing to the discharge during the period 1991-1993, indicating water intrusion into the cells.

Based on hydraulic gradients, different sources of water intrusion are possible. These are: the sea, the leachate, and the groundwater beneath the old landfill located to the southeast of the special cells. During the period 1991-1993, the average difference in piezometric head between the sea and the special cells was about 1.5 m, while the difference between leachate heads in the old landfill and the special cells was about 3 m. If an average monthly inflow of 16.5 mm is assumed, net infiltration plus extra inflow produces the graphs shown in Figures 3.2 and 3.3.

A lag time between infiltration and discharge of less than one month is suggested when weekly values are plotted. When net infiltration and observed discharge are compared on a monthly basis, a 'mirror effect' is observed. The leachate curve predicted by HELP is less undulating and shows a longer lag time for peaks when compared with field data. Model calibration to account for a faster response, by increasing hydraulic conductivity, reducing porosity and wilting point for the waste layer, produced no significant improvement in the match between observed and simulated discharge curves. Therefore, the best available curve to represent the measured discharge was still the curve for net infiltration.

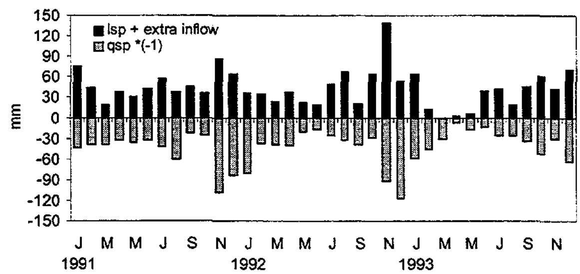


Figure 3.2. Uncovered special cells. Discharge (q_{sp}) and net infiltration (I_{sp}) plus extra inflow of water. $R^2 = 0.61$.

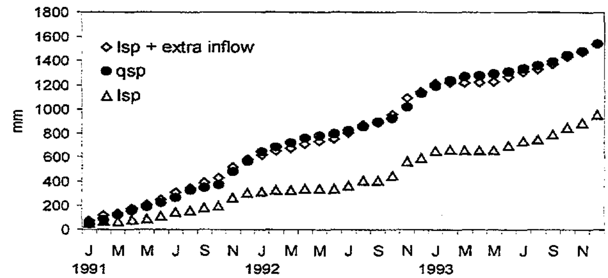


Figure 3.3. Uncovered special cells. Cumulative values (I_{sp}+ extra inflow) and q_{sp}:$R^2 = 1.00$. I_{sp} and q_{sp}: $R^2 = 0.99$.

Comparing discharges observed, with discharge estimated by the HELP model for capped cells, it was concluded that the covering layer produced a discharge reduction comparable to that obtained with an evaporative zone depth of 40 cm. After capping, a smoothing of the time distribution of the rainfall expressed as discharge was observed, and the correspondence between infiltration peaks and the discharge peaks disappeared.

Biocells

Depending on the MSW disposal rate, sludge infiltration rate, and weather conditions, instead of drying by evaporation, the sludge may contribute directly to the leachate discharge during the active filling period. This hypothesis is compatible with the water balance calculations from HELP modelling for biocells during the period 1991 to 1997 (Fig. 3.4).

Due to contradictory observations in the literature, further investigations in the laboratory and in the field are required.

The rheological properties of sludge from different sources, treatment processes and chemical composition are likely to differ and affect the fate of the sludge water once emplaced in MSW landfills. Of the rheological properties, those of importance are viscosity, de-waterability, capillary suction time, particle size distribution and drying characteristics. A general model to describe the fate of the original sludge moisture content following disposal, independently of its rheological properties, is probably not feasible.

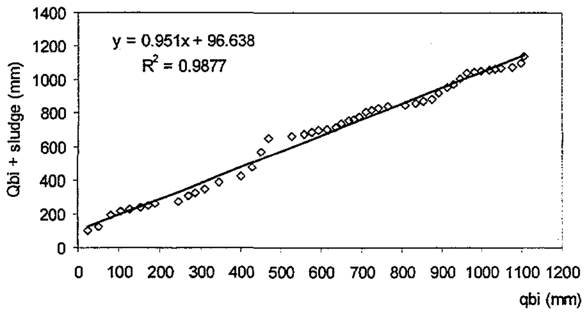


Figure 3.4. Capped biocells: regression curve of cumulative measured discharge (q_{bi}) and estimated discharge (with HELP) plus water from sludge (Q_{bi} + sludge). 1994-1997.

The water storage capacity available in MSW landfills during active filling (which could be increased due to sludge co-disposal) was not observed for special waste as a consequence of a smaller practical field capacity and/or more intense channelling, resulting in a high discharge being observed even during filling. Consequently, for uncovered special cells, the simple water balance curve of net infiltration may better represent the discharge than any leachate generation curve produced by the HELP model. Soil cover leads to a smoothing of the time distribution of the rainfall due to its hydraulic resistance, storage capacity, and higher evaporation loss, expressed as increased depth of the evaporative zone. This effect was observed after closure in both types of cells.

The HELP model can be used to simulate the hydrological performance of landfills with a top layer of high hydraulic conductivity (e.g., bare waste in fly-ash landfills), provided that precautions are taken to avoid model convergence problems. The addition of a hypothetical top layer in the landfill profile with hydraulic characteristics at the limit needed to obtain model convergence, is suggested.

The fact that special waste cells showed faster production of greater amounts of discharge, in response to infiltration, than that observed in MSW landfill cells must be considered when designing *on-site* leachate treatment systems for special waste landfills. The mentioned differences in hydrological behaviour are be expected to affect the pollutant transport associated with leachate generation mechanisms.

3.3 Stormwater Runoff and Pollutant Transport (Papers III and IV)

Runoff coefficients and initial abstraction based on field measurements and values described in the literature were used to estimate runoff generation at Spillepeng waste management park, where landfilling is one of 9 main waste handling practices.

The concentrations of different constituents were determined for eight areas and four roads during four rainfall events in a one-year period, by analysing a total of 48 samples. The activities carried out in the mentioned areas included: mechanical sorting and storage of slag; car park for cars and machinery; recovery yard and storage of compost soil; wood chipping and wood storage; composting or different types of organic waste and; storage of industrial and demolition waste.

Paper III - The main objectives addressed in were:

- to characterize the stormwater runoff at different areas using different waste handling activities;
- to compare the results of stormwater runoff from different industrial activities, with leachate generated at the same waste management park;
- to estimate the pollution load (expressed as pollutant load in mass per hectare per year), released by different activities; and
- to discuss the efficiency of infiltration elements in landfill sites for stormwater treatment.

Using nonparametric statistics, medians and confidence intervals of the medians for 22 stormwater quality parameters were calculated. Suspended solids, chemical demand of oxygen, biochemical demand of oxygen, total nitrogen and total phosphorus were determined. Runoff from several areas showed measured values above standard limits for discharge into recipient waters, even higher than those of leachate from capped landfill cells. Of the heavy metals analysed, copper, zinc and nickel were the most prevalent, being detected in every sample. Higher concentrations of metals such as zinc, nickel, cobalt, iron and cadmium were found in runoff from composting areas, than from areas containing stored and exposed scrap metal. This suggests that factors other than the total amount of exposed material affect the concentration of metals in runoff, such as binding to organic compounds (mainly dissolved organic carbon) and hydrological transport efficiency.

The median values of the concentration of several constituents from different areas and roads with their respective confidence intervals are presented in **Paper III.** The contribution to the annual pollutant load (kg/ha per year) from stormwater and leachate from different cells at Spillepeng landfill is shown in Fig. 3.5.

Most of the constituents analysed in the stormwater runoff generated at Spillepeng showed a wide confidence interval for the median of the concentration values, for sample size $N = 4$. A high variation in time is expected for stormwater runoff composition from active surfaces at waste management parks.

The primary reason is the frequent change in activities carried out on these surfaces, as well as changes in the amounts and characteristics of the waste handled or treated.

Sampling programmes with shorter time intervals and more samples are likely to increase data reliability and reduce the confidence interval.

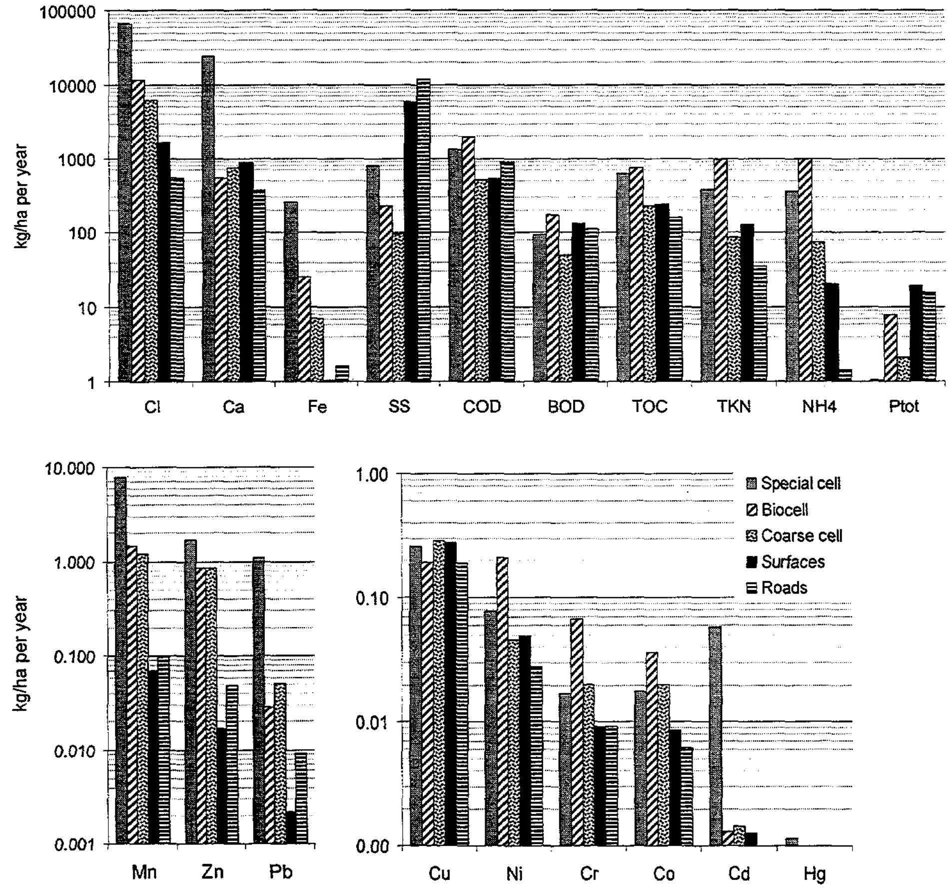


Figure 3.5. Pollutant load (kg/ha per year) transported by stormwater from areas and roads and transported by leachate from capped landfill cells, Stage I (1994-1998).

Hogland (1993) estimated that about 20 million m^3 of stormwater are discharged annually into The Sound, containing about 5% of the total pollutant load discharged along the Swedish coast, including some discharge from Spillepeng.

When the pollutant load discharged into The Sound is transformed into mean concentrations, the Spillepeng stormwater values corresponded to 21 times, 19 times, 1.6 times, 67%, 34%, 21% and 8% of the P_{tot}, N_{tot}, Cu, Cr, Zn, Cd and Pb discharged into The Sound. Therefore, the Spillepeng stormwater is more polluted than the average discharge into The Sound for the first three constituents (P_{tot}, N_{tot} and Cu), but not for the other four constituents (Cr, Zn, Cd and Pb).

At the Spillepeng site, it can be assumed that most rainfall will not produce surface runoff. This assumption is based on the precipitation pattern of southern Sweden, where about 71% of the rainfall events do not generate stormwater runoff from the main surface material used at Spillepeng (gravel), which covers about 69% of the active areas and roads.

Because most of the rain that falls is in small amounts, initial storage and infiltration of a small runoff amount for every rainfall event, as well as the first flush of larger rainfall events, are likely to prevent detrimental effects on surface water or groundwater to a large extent. However, due to the high concentration of pollutants found in the runoff generated from various areas and roads, which can even exceed leachate values, as well as the total pollutant load generated with runoff before infiltration, the stormwater management strategy for such sites as Spillepeng should be carefully considered.

The pollutant transported by stormwater represents a significant environmental threat, comparable to that of leachate. Careful design, monitoring and maintenance of stormwater runoff drainage systems and infiltration elements are required, if infiltration is planned to be used as an on-site treatment strategy.

There is an evident need for further hydrological evaluation and chemical characterization of runoff from waste management parks in order to properly address the effects of the runoff composition on treatment effectiveness by infiltration, and the time limit for which this treatment remains effective in different types of soil. Only after further studies can the best stormwater management practices for waste disposal sites be defined.

Paper IV - The main objective was:

- to compare the pollutant concentration ranges found in the stormwater runoff in these areas and from these roads, with the stormwater runoff from typical roadway, industrial and residential areas in Sweden.

Mechanisms governing accumulation, wash-off, and transport of different pollutants were not the scope of this study and were therefore only mentioned briefly. Management practices to reduce pollutant transport by stormwater runoff were suggested.

Field measurements and calculations based on a 30-year historical series of precipitation were used to derive runoff coefficients, initial abstractions for different surfaces (asphalt, gravel, steep slopes of landfill cells) and average annual stormwater runoff.

The transport of pollutants by stormwater runoff as it related to land use for traffic, residential, and industrial purposes in Sweden, was compared with that found at the

Spillepeng waste management park and the baled waste storage area at the Lidköping incineration plant.

Stormwater runoff from the study sites showed much higher concentrations of chemical demand of oxygen, total nitrogen and total phosphorus than typical concentration ranges for runoff from general roadway, industrial and residential areas. With respect to trace elements copper, zinc, and lead, the reverse occurred (Fig. 3.6).

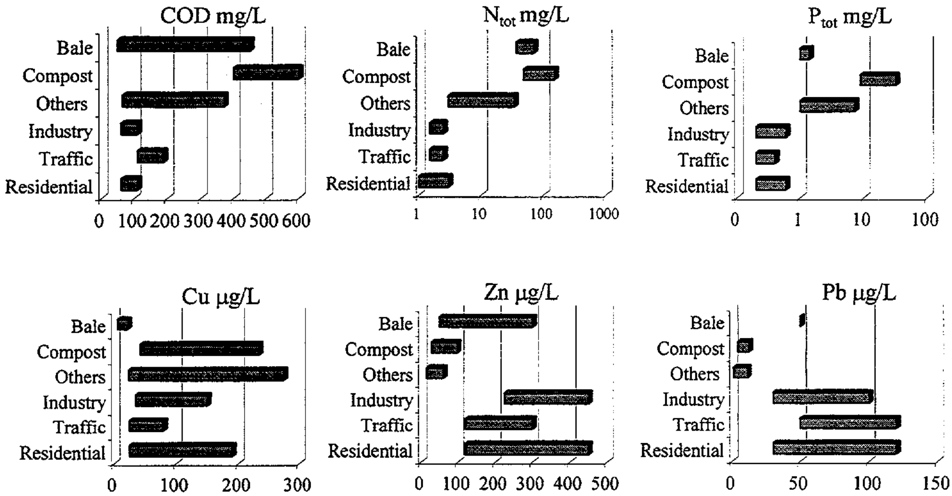


Figure 3.6. Concentration values in stormwater from storage of baled waste (Bales) at the Lidköping site; and composting areas (Compost), other areas and roads (Others) at the Spillepeng site. Typical Swedish ranges for Industry, Traffic and Residential areas with more than 50 inhabitants/ha (Malmqvist et al., 1994). Chemical oxygen demand (COD), total nitrogen (N_{tot}), total phosphorus (P_{tot}) (mg/L), Cu, Zn and Pb (µg/L).

Of all activities carried out at both sites, runoff from the composting area for garden waste exhibited the highest median concentration of iron, zinc, nickel, lead and cobalt. High concentration ranges for these heavy metals were found in runoff from urban areas from which the garden waste originates.

An explanation for this observation was derived using a material flow approach, assuming an efficient runoff transport mechanism for metals bound to organic compounds released with runoff from composting piles.

3.4 Statistical Approach to Groundwater Quality Time Series Analysis (Paper V)

Monitoring of groundwater quality beneath landfill sites usually aims to detect differences between up- and down-gradient locations and to identify short- and medium-term trends. In **Paper V,** the extent to which these goals can be achieved using different statistical approaches is addressed. The case study involved is the groundwater aquifer in a limestone cliff beneath Spillepeng landfill, sampled quarterly over a nine-year period (1991-1999). The main objectives were:

- to detect and estimate changes occurring in the quality of the groundwater in a limestone aquifer underneath the landfill site;
- to assess the adequacy of basic statistical methods for quarterly sampling;
- to verify the adequacy of the presently selected parameters and observation wells for the case study monitoring programme; and
- to suggest improvements in the monitoring programme based on the results obtained.

Six observation wells were installed in the limestone aquifer. The groundwater wells are in the following locations and order (Fig. 3.7):

- up-gradient of the old landfill (well G13);
- inside the old landfill, 320 m from well G13 (well G35);
- up-gradient of the landfill expansion, Stage I (wells G36 and G37); and
- down-gradient of Stage I (wells G41 and G42).

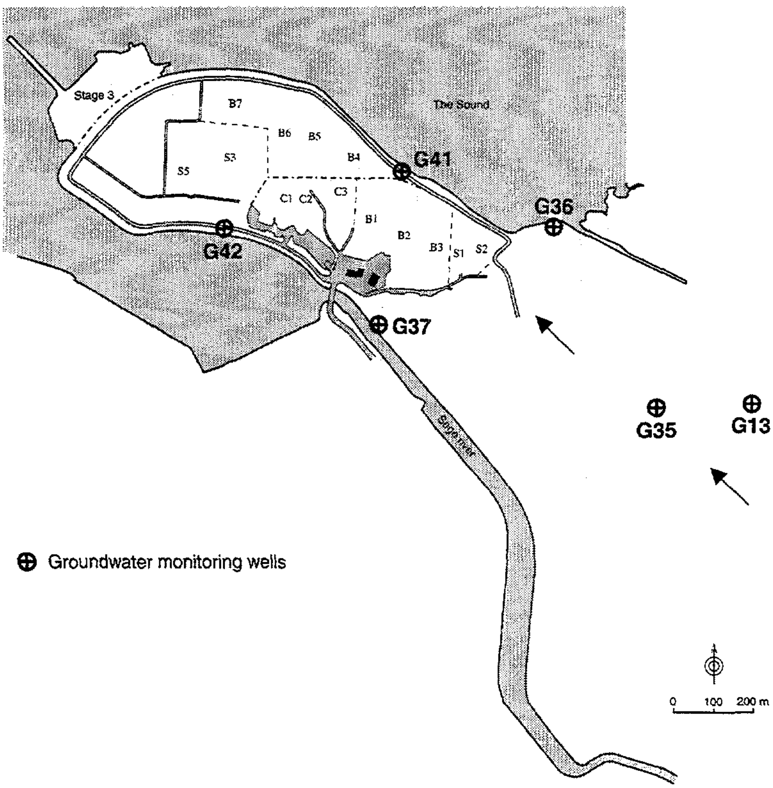


Figure 3.7. Spillepeng landfill. Groundwater monitoring wells G13, G35, G36 and G37 (old landfill); G41 and G42 (boundary between Stages I and II).

When this study was performed, no data were available from wells G36 and G37. The parameters monitored were: total nitrogen (N_{tot}), total organic carbon (TOC), chemical oxygen demand (COD), electric conductivity (EC), chloride (Cl), and pH.

The results of the exploratory statistical analyses indicated that groundwater quality time series data sets were generally not normally distributed. The analyses showed they were serially correlated, with seasonal effects in 39% of the cases, with outliers in 11 out of 22 time series sets; and with censored data (i.e., below analytical detection limit) for COD in the up-gradient wells G13 and G35. These results suggested that in principle, nonparametric methods are the most appropriate to estimate spatial differences and to investigate trends over time.

Statistical Tests

Correlation. To investigate the correlation between pairs of time series data sets, the nonparametric Kendall rank correlation test was compared with the Pearson correlation test. The autocorrelation test was performed with the autocorrelation function (acf) for different time lags.

Differences between pairs of wells. To detect differences in pollutant concentrations in down-gradient wells compared to up-gradient wells, the nonparametric rank-sum test was compared with the paired *t*-test (paired samples). The magnitude of the differences was estimated with a Hodges-Lehmann estimator and the difference of means, respectively.

Trend. For trend analysis, both nonparametric tests, the Mann-Kendall test and seasonal Kendall test, were carried out. To estimate the magnitude of the trends, the Sen's slope estimator and the Seasonal estimator were used, respectively.

Forecasting. Forecasting was performed by applying the Sen's slope estimator, the Seasonal slope estimator and Winter's method of forecasting for time series data sets, for those parameters that showed a statistically significant upward trend, excluding those parameters where forecasting may produce unrealistic values (e.g., pH).

Differences between down-gradient wells and up-gradient wells

The pairs of wells that showed values significantly different to each other, based on the rank-sum test, are shown in Table 3.1.

The fact that well G35 showed significantly lower values for EC and TOC compared with up-gradient well G13, is viewed with caution. Other aquifer characteristics may contribute to this result. Monitoring at wells G35 and G37 (the latter is not included in the groundwater quality monitoring programme) has shown anomalously low and high average piezometric heads, respectively. These anomalous heads may be a result of vertical layering or fissuring in the limestone aquifer, which are being measured at slightly different depths within the aquifer.

Table 3.1. Significant differences ($\alpha = 0.05$) according to the rank-sum test.

Pair of wells (down-gradient: up-gradient)	Parameters
G35 > G13	pH
G41 > G13	pH
G42 > G13	pH, EC, COD, N_{tot}
G41 > G42	N_{tot}
G41 < G42	pH
G35 < G13	TOC, EC

Differences in groundwater quality may result from aquifer heterogeneity. Natural variations in aquifer geochemistry and hydraulic properties at different depths may account for such differences.

According to the groundwater flow contour lines, even if both wells, G41 and G42, are at the same average water piezometric head, the groundwater travelling distance beneath the landfill area to reach well G41 is much greater (1,100 m) than the travelling distance to reach well G42 (200 m). Therefore, higher concentration values for pollutants at well G41 are expected if contamination by the landfill is confirmed.

Trend analysis

The constituents that presented significant upwards or downwards trends in any of the tests performed are shown in Table 3.2.
For trend analysis, the results found for N_{tot} in wells G13 and G35 again require additional discussion.

Table 3.2. Trends that are statistically significant ($\alpha = 0.05$) according to the Mann-Kendall test or the Seasonal Kendall test for trend.

Monitoring well	Upwards trend ↑	Downwards trend ↓
G13	-	N_{tot}, COD
G35	pH, EC	-
G42	N_{tot}, EC, Cl	pH
G41	N_{tot}	pH, Cl

A decrease in N_{tot} detected in well G13 should also appear in well G35, located about 600 m from G13, unless there was a simultaneous increase of N_{tot} in the groundwater underneath the old landfill, maintaining the level in G35; or the discrepancies reflect aquifer heterogeneity, as discussed previously.

The results observed for chloride and conductivity at wells G41 and G42 may be misleading, due to the "step-type" trend observed in the beginning of the time series.

These were not excluded from the trend analysis, since the causes of such effects are not clear. The possible influence of seawater on the chloride content of groundwater at the shoreline means these parameters are not suitable for monitoring groundwater contamination by leachate.

The Seasonal Kendall test was more powerful than the Mann-Kendall test in detecting trends and was more robust against the effect of outliers.

Forecasting

Forecasting of N_{tot} at well G41 using Winter's method is shown in Figure 3.9.

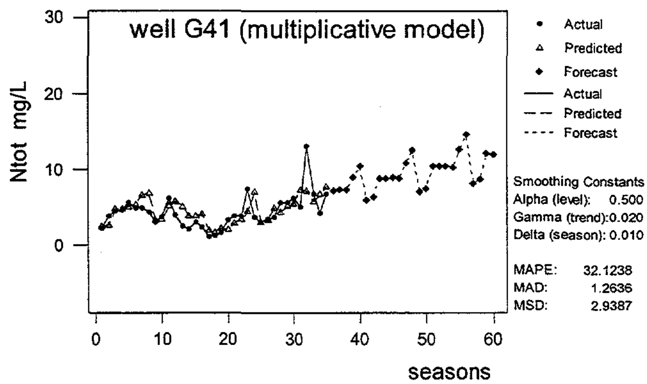


Figure 3.9. Forecasting by Winter's method. Measures of accuracy: Mean Absolute Percentage Error (MAPE), Mean Absolute Deviation (MAD), Mean Square Deviation (MSD).

Winter's method, as well as the Sen's slope and Seasonal slope estimated by a Mann-Kendall test and Seasonal Kendall test, respectively, were applied to forecast the time required to reach a concentration of about 15.5 mg/L for N_{tot} in the groundwater. This value represents the sum of:

- the mandatory limits of 50 mg/L of nitrate as NO_3 (equivalent to 11.3 mg/L NO_3-N); plus
- 4 mg/L of ammonia as NH_4 (equivalent to 2.5 mg/L NH_4-N) in potable water according to the European Commission (Gray 1999); plus
- the organic nitrogen content estimated by using the average ratio of organic nitrogen/N_{tot} (0.11) found in the leachate from Stages I and II during 1991-1999.

Due to the long time period used for forecasting, these values should be considered to have a great degree of uncertainty. However, this illustrates the differences obtained by use of different non-parametric and parametric slope estimators.

Moreover, the time to reach these limits would be much shorter, or may already have been reached, if, for instance, the N_{tot} measured were mainly composed of ammonia.

Table 3.3. Estimation of the time (in years) necessary to reach the European Commission mandatory limits for nitrate NO_3-N and for ammonia NH_4-N in potable water (beginning at December 1999).

N_{tot}=15.5 mg/L	Sen's slope estimator	Seasonal slope estimator	Winter's method
well G41	43 years	7 years	11 years
well G42	72 years	13 years	16 years

The possibility of an upward or downward trend reflecting simply local effects due to aquifer heterogeneity rather than a real degradation or improvement of the groundwater quality, should also be considered.

Nevertheless, for improved quality control, it is suggested that the individual nitrogen-containing parameters nitrate, nitrite, and ammonia, as well as a suitable organic compound for monitoring groundwater contamination by landfills, such as phenol, be included in future groundwater monitoring. The inclusion of two extra groundwater wells (G36 and G37) in the monitoring programme is also suggested to achieve a better overview of the groundwater quality changes over time and space. Caution should be taken during expansion of the landfill, and the use of a synthetic liner should be considered.

4. CONCLUSIONS

The conclusions can be summarized in seven concluding remarks.

1. Similar to the mechanism described in the literature for stored forest fuel material, natural convection plays an important role in the degradation process of waste fuel material by supplying oxygen and releasing heat.

2. Waste stored in loose piles is not a recommended method for storage of waste as fuel, particularly for periods longer than one month, due to loss of energy content and risk of self-ignition. An alternative that achieves more homogeneous and lower permeability within the waste mass by powerful compaction, such as baling, is preferable.

3. Landfill cells that contain predominantly fly ash and other special waste, such as sheets of asbestos and cement, contaminated soil, blasting grit, rest fraction of sorted slag and clinical waste, showed a practical volumetric field capacity as low as 0.066, hydraulic conductivity as high as $1.0 \cdot 10^{-2}$ cm s^{-1}. Due to these characteristics, a faster discharge response to rainfall infiltration compared to MSW landfill cells is observed. This feature must be considered when planning and designing leachate management systems for special waste landfills.

4. Stormwater runoff from essentially all activities and areas at waste management parks was found to be an important source of pollutants, particularly with respect to the chemical demand of oxygen and nutrients. Runoff from compost area for garden waste, and the area for mechanical sorting and storage of slag plus runoff from other land urban uses, such as traffic, residential and industrial areas, are likely to be more important sources of heavy-metal water pollution.

5. Runoff from composting areas showed the highest concentrations of heavy metals of all areas and roads at the waste management park, even higher than the area containing stored and exposed scrap metal. This suggests that factors other than the total amount of exposed material, such as binding to organic compounds and hydrologic transport efficiency, play an important role in determining the concentration of metals in runoff.

6. The detection by statistical analyses of differences between up- and down-gradient monitoring wells, as well as trends in groundwater quality, which are not otherwise detected by simple inspection of graphic plotting of the time series data sets, demonstrates the need for proper statistical support during water quality monitoring. The analysis and interpretation procedures for monitoring data, are not addressed in the Annex III of the new EU Directive for landfill (Annex III

Control and Monitoring Procedures, Council Directive 1999/31/EC, 1999) and should be further discussed.

7. However, before final conclusions about trends may be derived, aquifer geochemistry and aquifer heterogeneity must be considered, since an upward or downward trend may reflect local effects, rather than a real improvement in, or degradation of, the groundwater quality.

Water, gas and heat transport within waste fuel storage and self-ignition

The degree of compaction achieved by proper procedures (for each 50 cm of waste, a 32-tonne compactor passes over it four to five times), when waste piles are constructed for storage purposes, cannot eliminate the risk of self-ignition. The permeability distribution in the pile, channelling and natural convection create zones with sufficient moisture and oxygen supply to promote biodegradation, leading to heating and spontaneous combustion in both compacted and uncompacted piles.

Considerable changes in temperature and gas composition occurred during the first weeks of waste fuel storage, reflecting the intense biological activity taking place in the readily-biodegradable fraction of the waste. This raises the temperature into the optimum range for chemical oxidation reactions (65 to 75 °C). At this point, in the presence of catalytic materials, such as metals, optimum conditions for self-ignition and spontaneous combustion are reached.

Together with the waste's own weight, increasing participation by non-biological chemical processes in the degradation process, and reduction of energy content of the waste, promote settling. Similarly to landfills, regions with different permeability are expected to develop. The permeability distribution in the pile will govern the major pathways for airflow in the pile, and which, due to higher heterogeneity, is expected to be higher than in stored forest fuel material.

Some practical countermeasures to reduce the risk of fire are: the removal of coarse and readily-degradable waste; the avoidance of metallic materials; ensuring piles are lower than 5 m in height; construction of piles parallel to the direction of the prevailing wind; protection of piles from strong winds by wind barriers. However, due to the observed risk of self-ignition and spontaneous combustion of storage in loose piles, baling of waste fuels, while still dependent on further investigation, seems to be a more promising strategy.

This procedure prevents self-ignition, as the biodegradation rate is reduced with no temperature increase. The superiority of the baling technique over the loose piles as regards the aspects mentioned above, is probably a consequence of the higher degree of compaction achieved by the former.

Water flow throughout unsaturated medium

The water storage capacity available in MSW/sludge landfill cells (biocells) during active filling was not observed for special cells as a consequence of a smaller practical field capacity, resulting in a high level of discharge being observed, even during filling. Consequently, for uncovered special cells, the simple water balance curve of net infiltration, may better represent the discharge than any leachate generation curve produced by the HELP model. Soil cover leads to a smoothing of the time distribution of the rainfall due to its hydraulic resistance, storage capacity, and higher evaporation loss, expressed as increased depth of the evaporative zone.

The HELP model can be used to simulate the hydrological performance of landfills with a top layer of high hydraulic conductivity (e.g., bare special waste in active landfills), provided that precautions are taken to avoid model convergence problems. The addition of a hypothetical top layer in the landfill profile with hydraulic characteristics at the limit needed to obtain model convergence is suggested.

The differences suggested by this study between biocells and special cells regarding water flow though the waste mass, indicate that diverse leaching behaviour can also be expected due to the association between water flow mechanisms and pollutant transport.

Depending on the MSW disposal rate, the sludge rheological properties, the sludge infiltration rate and weather conditions, instead of drying by evaporation, the sludge may contribute directly to the achievement of the practical field capacity and to the leachate discharge during the active filling period. Due to contradictory observations in the literature regarding the fate of the sludge moisture once disposed of in landfills, further investigation in the laboratory and in the field is required. This specific issue is important, particularly for non-EU countries, where co-disposal is expected to continue as one of the most common strategies of sludge handling.

Stormwater runoff and pollutant transport at waste management parks

Of 22 different parameters analysed in stormwater runoff from areas where several waste handling techniques are carried out, and from roads in the Spillepeng waste management park, suspended solids, chemical oxygen demand, biochemical oxygen demand, total nitrogen and total phosphorus showed measured values above standard limits for discharge into recipient waters. For some areas at Spillepeng, these values were even higher than those of leachate from covered landfill cells.

Of the heavy metals analysed, copper, zinc and nickel were the most prevalent, being detected in every sample. Higher concentrations of metals, such as zinc, nickel, cobalt, iron and cadmium, were found in runoff from composting areas than in areas containing stored and exposed scrap metal, suggesting that factors other than the total amount of exposed material, such as efficient transport mechanisms (e.g.

binding to organic compounds, particularly dissolved organic carbon) affect the concentration of metals in runoff.

By comparison with stormwater runoff from other land uses, such as urban residential areas, general roadways and industrial areas in Sweden, stormwater runoff from waste management parks was confirmed to be an important source of pollutants, particularly with respect to the chemical demand of oxygen and nutrients, such as phosphorus and nitrogen. Conversely, excluding high concentrations of copper in runoff from composting of garden waste and storage/sorting of slag areas, runoff from other land uses has shown higher concentrations of heavy metals than runoff from the waste management site.

Because most of the rain that falls in southern Sweden is in small amounts, its storage and infiltration, as well as storage and infiltration of the first flush of larger rainfall events, are likely to prevent the detrimental effects of discharging pollutants in surface water or groundwater to a large extent. However, due to the relatively high concentration of some pollutants found in the runoff, careful design, monitoring and maintenance of stormwater runoff drainage systems and infiltration elements are needed if infiltration is to be used as an on-site treatment strategy.

There is an evident need for further hydrological evaluation and chemical characterization of runoff from waste management parks in order to properly address the effects of the runoff composition on treatment effectiveness by infiltration, and the time period over which this treatment remains effective in different types of soil.

Statistical approach for groundwater quality time series in monitoring programmes

The groundwater time series data sets from Spillepeng landfill were, in general, non-normally distributed, as evidenced by serial correlation, seasonal effects with outliers, and censored data (data below analytical detection limit). Non-parametric procedures are therefore the most appropriate for statistical analysis of these data. The non-parametric rank-sum test to detect differences was indeed more powerful and robust than the *t*-test. The seasonal Kendall test was more powerful than the Mann-Kendall test.

The total nitrogen downwards trend at up-gradient well G13, and the upwards trend at G41 and G42, the most down-gradient wells, may reflect groundwater improvement up-gradient in the landfill and groundwater contamination by the landfill, respectively. However, indications of the existence of aquifer geochemical heterogeneity suggest that an upward or downward trend may reflect local effects, rather than a real improvement in, or degradation of, the groundwater quality.

Nevertheless, it is suggested that measurement of nitrate and ammonia forms of nitrogen, as well as phenol, be included in the monitoring programme. The observation wells (G36 and G37) used so far for water head measurements should also be included in the quality monitoring programme. Extra groundwater protection is suggested, and the use of a synthetic liner during expansion of the landfill, ought to be considered.

5. REFERENCE LIST*

* This list includes all papers referred in the introductory part of the thesis and/or in the appended papers.

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