ERNATIONAL HYDROLOGICAL PROGRAMME

Proceedings International Workshop on Impact of Waste Disposal on Groundwater and Surface Water

August 15-19, 1988 Copenhagen Denmark

Convened by the Danish National Water Council
and organized in cooperation with UNESCO, IAH. and NAEP

Miljøstyrelsen

Editors: Erling Rørdam and Steen Vedby

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PREFACE

The International Workshop on **Impact of Waste Disposal on Groundwater and Surface Water** was held 15th - 19th of august 1988 ar Schaeffergaarden, Copenhagen, Denmark.

The present publication contains the Proceedings of this Workshop, including most of the submitted papers. The proceedings consist of five main parts;

Part I : Recommendations and opening adress,
Part II : Processes within the disposal sites,
Part III : Environmental aspects and cases,
Part IV : Remedial actions and hydrological controls,
Part V : Strategy, forecasting and policy.

The workshop was a contribution to the International Hydrological Programme (IHP) - phase III under United Nations Educational, Scientific and Cultural Organization (UNESCO). For further introductory and background information about the workshop, the reader is referred to the Introduction and the Recommendations.

The workshop was organized by the Danish National Water Council and the National Agency of Environmental Protection of Denmark in cooperation with UNESCO and the International Association of Hydrogeologists (IAH).

The technical programme and technical excursion during the workshop was arranged by following organizing committee:

Erling Rørdam, Chairman for the Danish National Water Council, National Agency of Environmental Protection (Chairman)

Steen Vedby, National Agency of Environmental Protection, (Organizer and technical secretary)

Thomas H. Christensen, Department of Environmental Engineering, Technical University of Denmark,

Karsten Høgh Jensen, Institute of Hydrodynamics and Hydraulic Engineering, Technical University of Denmark,

Leo B. Langgaard, Water Quality Institute,

Henrik Kærgaard, N & R Consult - Terraqua,

Max Jensen, I. Krüger Ltd.,

Ole Stang, Kemp & Lauritzen Ltd.,

Lars Jørgen Andersen, Geological Survey of Denmark, International Association of Hydrogeologists,

Tove Bjerregaard, National Agency of Environmental Protection, (secretary), and

W. H. Gilbrich, UNESCO´s Division of Water Sciences.

During the workshop a drafting committee was established to produce the final recommendations. Following participants were members of this committee:
Ronald Hill, U.S.A., **R.E. Jackson**, Canada, **M.S. Bedinger,** U.S.A., **Peter Kjeldsen,** Denmark, and **Steen Vedby,** Denmark.

We wish to thank **UNESCO, IAH. NAEP** (Denmark), the organizing committee, the drafting committee and the participants for their excellent performance and support.

Copenhagen, November 1988

Steen Vedby

Erling Rørdam

INTRODUCTION

The present publication contains the papers presented to the Copenhagen Worjshop as well as recommendations elaborated by the workshop participants. The workshop was very much concerned by the menace which threatens present and future generations through the long-lasting deterioration of surface water bodies and aquifers by waste. The proceedings are meant to draw world-wide attention to the problem and to offer solutions. The Danish Water Council and the National Agency of Environmental Protection decided, therefore, to publish and widely distribute the workshop proceedings as a contribution to UNESCO's International Hydrological Programme (IHP).

The third phase of the International Hydrological Programme covers four main fields: hydrological processes and parameters for water projects, the influende of man on the hydrological cycle, retional water resources assessment and management, and education, training and public information. During the history of the IHP, emphasis has shifted from computational water quantity hydrology to the assessment of man's influence on the hydrological cycle and water quality aspects have been increasingly taken into consideration.

The section og the third phase of the IHP on the influence of man encompasses three major themes. Theme 6 deals with the methods for assessing the changes in the hydrological regime due to man's influence; Theme 7 concerns environmental impact studies of water projects while Theme 8 deals with specific influence of man on the hydrological regime.

Within Theme 8, four projects have been created. One collects and evaluates the information on the effects of agriculture and forestry practices; one is related to urban hydrological processes and one deals with the impact of acid precipitation on the hydrological and ecological systems.

Project 8.2 is devoted to hydrological aspects of liquid disposal by industry and mining, waste disposal on land and radioactive disposal. The liquid disposal has been the subject of the international workshop "Metals and metalloids in the hydrosphere, impact through mining and industry, and prevention technology" held at Bocum (Federal Republic of Germany) in September 1987. The proceedings will be prepared by the IHP/OHP (National Committee of the Federal Republic of Germany) and published by UNESCO in 1988. The radioactive aspects have been treated by IHP rapporteur, Mr. M.S. Bedinger (USA), in the UNESCO publication "Hydrological aspects of land disposal of radioactive waste" (1987). The hydrological aspects of solid waste disposal on land have been discussed at an international workshop organized by the Danish IHP National Committee at Schaeffergaarden/Copenhagen, from 15th to 19th August 1988, vo-organized by UNESCO and the IAH and with the financial assistance of UNESCO.

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Recommendations and opening address

PREAMBLE

The workshop on "Impact of Waste Disposal on Groundwater and Surface Water" meet to discuss primarily the hydrogeological aspects of waste disposal and engineering solutions to site remediation. In addition priority ranking scheme for site investigation and remediation, risk assessment and strategies for future decision making among waste remediation options at the indutrialized as well as the developing nations were discussed.

The concensus that developed was that:

I. Prevention of contamination is clearly preferable to remediation. This assumes that cleaner technologies should be developed and all societies maximize seperation and recycling.

II. Waste problems should be solved by the creators of the problem on a timely basis and not passed on to other countries and generations.

III. Important solution to the hazardous waste problem is the collection, isolation and recycling of these waste at their source and their destruction at the source or centralized hazardous waste treatment facility.

IV. The municipal waste management problemn can largely be solved through public education and cooperation in the seperation and recycling of their waste.

V. In all these matters vigorous political leadership is rewuired.

Recommendations.

Given the success of the present workshop on defining the problems and their solutions is recommended:

I. We encourage futher international exchange of information and ideas between industrial, newly industrial, and developing countries.

II. In order to facilitate this exchange and to meet the environmental goals set out in our preamble we recommend the creation of an international working group in hydrological aspects of waste control, that is, waste site investigation, design, remediation, and monitoring.

Specific topics to be considered:

1. The design and operation of hydrological monitoring systems.

2. The hydrological assessment of groundwater contamination at sites, with particular reference to fractures bedrock systems.

3. The development of guidelines to determine courses of remedial action, i.e. whether to take no action or to contain or to decontaminate the waste site.

4. The design of landfill caps, covers and capillary barriers.

5. The design and operation of purge well networks in relation to pump and treatment systems.

6. The integration of hydrochemical results in risk assessments and in remedial action ranking systems.

OPENING ADRESS

by Jens Kampmann, director general,
The National Agency of Environmental
Proection, Denmark.

It is a great pleasure for me and a great honour to be able to welcome such a highly qualified gathering of hydrologists to this international workshop in Copenhagen on the subject of "Impact of waste Disposal on Groundwater and Surface Water".

This workshop - which has been arranged by the Danish National Water Council which is also the Danish IHP Committee, and by the National Agency of Environmental Protection - is a contribution to UNESCO´s International Hydrological Program, to which Denmark has contributed actively over the years. The IHP is very highly regarded in Denmark because it has increased scientific knowledge and promoted research and cooperation in the field of hydrology throughout several decades. At the same time the program offers the possibility of closer cooperation and exchange of knowledge across national boundaries.

The theme of the workshop, which will focus in particular on the threat of chemical waste dumps to the aquatic environment, occupies a central place in Danish environmental policy, and much effort and many economic resources are being devoted to this problem at the present time.

On a global scale, too, the subject is of the greatest importance, and the extent of the problem has recently been emphasized by the Bruntland Commission.

With regard to hazardous waste (in other words, chemical waste) the Commission´s report on environment and development states that "the primary political goal must be to reduce the quantitites of waste that are being produced, and to transform an increasing proportion of it into resources for use and re-use". Only in this way will it be possible to reduce the quantities of chemical wastes which otherwise will have to be treated or disposed of by incineration, land deposition, or dumping at sea. Hazardous wastes are the cause of some of the most acute environmental problems facing us - primarily in the "old" industrialized countries - today.

However, there is another problem which recently industrialized countries and developing countries must consider seriously, otherwise rapid industrialization will quickly create the same acute problems, and that is, how to deal with hazardous waste.

It has been estimated that the industrialized countries produce about 90% of the world´s hazardous waste, and in 1984 alone between 325 and 375 million tons were produced. The threat to the environment is present wherever such wastes are produced, transported, disposed of, or destroyed.

Recognition of the problem:

It is only in recent decades that we have recognized that the production and disposal of chemical or other wastes creates a pollution problem.

Two hundred years ago no intensive production of hazardous chemicals took place at all, and the quantities of waste were negligible. By way of interest I can tell you that the offices of the National Agency of Environmental Protection are built on landfill and waste from the city of Copenhagen, deposited here since the 16th century.

With the onset of industrialization the quantities of waste increased. However, the production and deposition of hazardous waste is mainly a problem from the thirties and the period since the Second World War.

Until the mid-sixties the restrictions on production were slight, and waste was disposed of on uncontrolled, open landfills - often disused gravel pits. Since the beginning of the seventies the environmental restrictions have been strengthened - also in Denmark, where in 1973 the Ministry of the Environment was created and an Environmental Protection Act was passed. This law included the requirement that from 1974 onwards all land deposition should take place on approved waste disposal sites.

However, the threat to the environment by chemical waste was not generally recognized until the late seventies and early eithties. It is notable that most countries first started work on the problem after a big "case".

This happened in Denmark after a pollution problem in 1980 near a Danish chemical factory in North-west Jutland, where the environmental studies and clean-up costs amounted to about 80 mill. Danish Kroner (12 mill. dollars). In the same way, a case on Love Canal gave the stimulus for more widespread surveys and clean-up programs in the USA.

The Jutland case and a couple of others surfaced soon afterwards, led in 1982 to a survey of all waste disposal sites in Denmark. The survey covered about 3000 sites, of which about 500 were thought to need further action because of chemical waste. In 1983 the Danish Parliament allocated 400 mill. Danish Kroner to a law on waste sites, which empowered the government to carry out investigations and remedial action at waste sites contaminated the chemical waste.

Since then, about 100 sites have been studied, and remedial actions have been necessary at about 30. In addition to this, a similar number of surveys and remedial actions have been carried out on a voluntary basis in accordance with the Environmental Protection Act.

Experience from this work, and from a great number of relevant research and development projects, means that we in Denmark are now much better equipped to deal with water pollution problems in the future. At the same time, however, our improved knowledge and new surveys of waste sites and industrial properties have shown that a number of cases needing positive action are much greater than originally supposed. We know that this is also the case in other industrialized countries that we cooperate with.

It is therefore necessary to carry out research into techniques for survey and remedial action so that these problems can be solved in an efficient and cost-effective way.

In this connection I may add that the Bruntland report quotes the following figures for such programs: 10 bill. dollars in the FRG, more than 1.5 bill. in Holland, 20-100 bill. dollars in the USA. A figure of 60 mill. dollars (1986 value) is quoted for the Danish effort in coming years. However, a new study has recently reported that Denmark will probably need to use about 750 mill. dollars.

The enormous cost of remedial action makes it clear that in future, disposal of hazardous waste must be carried out in an environmentally acceptable way, for example by emphasizing recycling and better waste planning. Reduction of volumes of waste can also be achieved by adopting deposit/return systems and by taxing waste disposal.

This is also relevant for developing countries, where numerous problems have to be faced. Frequent heavy rainfall in tropical regions washes waste down into the ground below landfills, or causes overflow. Since wastes have rarely been pretreated, if at all, this leads to pollution of water supplies or to direct exposure of local populations to the overflowing wastes. Landfills are often sited close to industrial sites, which are often surrounded by poor residential areas or slums. Dangers of this type underline the need for planning of land use in developing countries, and the even greater need to realize and maintain such plans.

The developing countries themselves will come to bear the costs of inappropriate industrialization, and the final responsibility for ensuring sustainable development rests on each government. They must define their own environmental and developmental objectives, and choose clear priorities amongst the many demands that are made to their few resources. They will also need to seek more effective self-help methods for industrial and technological development.

It is my hope that this IHP workshop will give developing countries an overview of the nature of the problems, and the actions that are necessary to reduce the environmental consequences. At the same time we all hope for continued cooperation with our existing partners. I can add here that NATO has initiated a pilot program entitled "Demonstration of Remedial Action Technologies for Groundwater", which Denmark is contributing to. In addition, a major Danish research project, "Reclamation of Landfill Leachate Polluted Groundwater", forms part of a large EEC research program. We have great hopes of these programs.

I believe that we can conclude:

- that wastes and waste disposal are a steadily growing problem for industrialized countries
- that this problem needs intensified research and development of remedial measures
- that in the long term the problems can only be solved by adopting cleaner technologies and recycling
- that in the short and medium term we must dispose of hazardous waste by destroying toxic chemicals.

Denmark will continue to favour international cooperation in this field, for example through export of systems, participation in similar activities within UNESCO and similar organisations, and international development programmes.

With these words I would like to wish the participants a fruitful workshop, and a pleasant stay in Copenhagen, and hereby declare the workshop open.

Jens Kampmann
Director-general

Processes within the disposal sites

DEGRADATION WITHIN WASTE DISPOSAL SITES

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Summary

The under standing of microbial degradation processes in landfills is important for evaluating the pollutional potential and the possibilities for reducing the pollutional potential by technological operations.

The waste is passing different phases: in the landfill a short aerobic phase, an anaerobic acid phase a transitional phase ending in the stable methane phase. The most important environmental factors are pH, temperature and the water content in the waste.

Different techniques for enhancing the degradation process have been tried. Many of the techniques have been suceesfull. Especially recirculation of leachate is able to enhance degradation. Landfill are complex and inhomogeneous and a quantification of the effects of different enhancement techniques is not possible.

INTRODUCTION

In many places both in industrialized and developing countries, solid waste is not further treated before the final disposal at the waste disposal site. Codisposal of chemical waste at the landfills was formerly the common way of disposing chemicals in industrialized countries and still some codisposal is taking place. In many developing countries most of the chemical waste is probably disposed on landfills.

Many landfills especially in the past have not been lined with a clay or plastic liner to protect the underlying groundwater. Still many landfills are constructed without any sort of groundwater protection liner.

Waste disposal is due to these facts a world spread threat to local groundwater resources.

Especially the organic fraction of the mixed waste consist a threat to groundwater. The understanding of the degradation of the organics within the waste disposal site is an important task to

- evaluate the pollution potential of a landfill site now and in the future
- enhance the degradation processes by technological operation in this way decreasing the pollution strength and the time span of environmental impact of the landfill
- increase the production of methane gas if a gas utilization is considered.

In the following the generel degradation processes and the degradation of codisposed chemical wastes landfills are described. Important environmental factors and finally enhancement technology are evaluated.

DEGRADATION PROCESSES

Degradation phases

A mass of mixed solid waste which is disposed at a given time will pass several degradation phases:

I Aerobic phase.

The easiest degradable compounds are degraded aerobically by the oxygen originally present in the waste and by oxygen diffusing into the waste. The oxygen is used up and carbon dioxide is developed. The temperature will increase radically. The duration of this phase is in comparison to the following phases very short (ended within weeks)

II Acid phase

Oxygen is totally absent and strong anaerobic conditions are developing. Easily degradable compounds is degraded to fatty acids and carbon dioxide. Ammonia is formed by the degradation of proteins. The strong formation of acids leads to a decrease in pH. The duration of this phase is extremely depended of the environmental conditions present (durations from few months to several years have been observed.)

III Transitional phase

The formation of methane by degradation of fatty acids is starting (if the right environmental conditions are obtained). Also the reduction of sulphate to sulphide is starting. The reduced fatty acid concentration in the leachate leads to higher pH values.

IV Methane phase

The formation of methane is stabilized leading to fairly constant methane content in the landfill gas (appr.50 percent) In this phase the fatty acids produced, are nearly instantaneously degraded to methane and carbon dioxide leading to very low fatty acid concentrations in the leachate. The duration of this phase is normally high (normally measured in decades).

This picture given is of course very idealised. In a natural landfill the different parts have different ages, giving a mixed and more complicated picture.

The methanogenic environment

The methanogenic phase is as shown the longest and therefore the most important phase. This phase is a complex and sensible interplay between different groups of bacteria as shown on Figure 1 where the squares are the substrates and end product, the ellipsis the microbial groups and the arrows the reactions.

Organic matter

The composition of the organic matter in leachates is very complex but generally the content of humic substances is relatively high and the fatty acid contents low in methane phase leachates. The humic substances are heavily degradable.

Codisposed organic chemicals mainly consist of oil waste (where the most soluble fraction is the aromatic hydrocarbons such as benzene, toluene, xylenes, ethylbenzene etc.) and solvents where halogenated compounds such as trichloroethylene, trichloroethane etc. is commonly used. The degradation of organic chemicals within the disposal site is enhanced by the presence of the general organic content in the leachate. The general organic content sustains the biomass as the primary substrate, while the chemicals are degraded as secondary substrate at a higher rate than if the primary

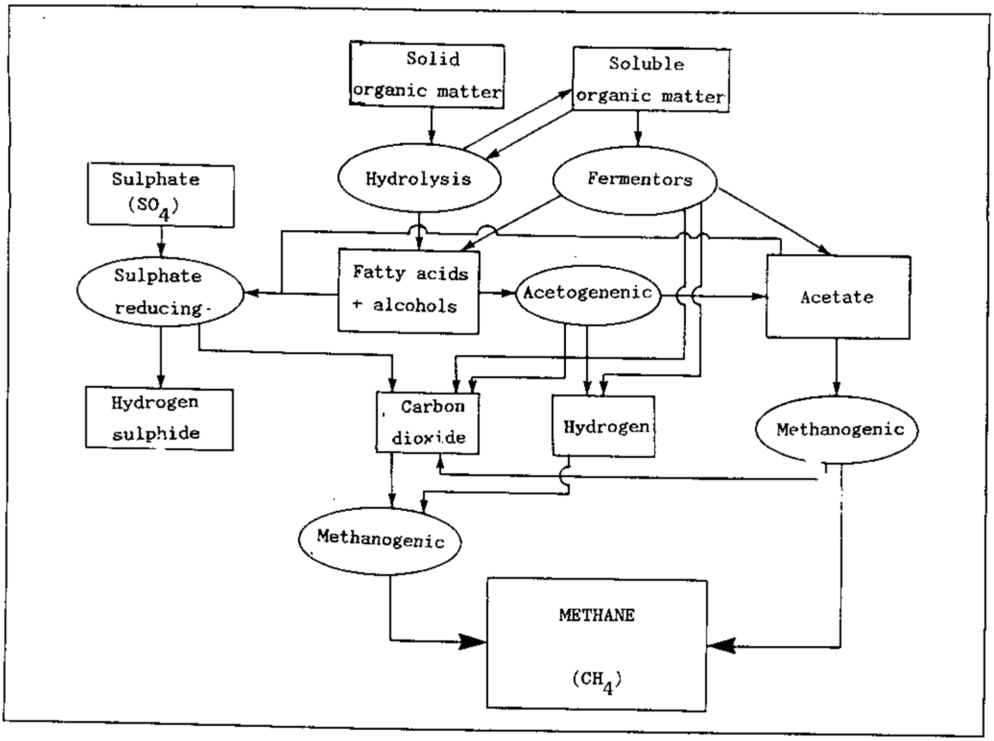

Figure 1: The metabolishm of substrates and the bacteria (only the most important substates and bacteria are shown).

substrate was absent. However, the aromatic hydrocarbons seems not to be degradable under methanogenic conditions. The halogenated compounds is anaerobically degradable by the "dehalogenation" process. In this process the chloroatoms are "clipped off" one by one. for instance is tetrachloroethylene degraded to trichloroethylene, dichlorophenols to monochlorophenols etc.

ENVIRONMENTAL FACTORS

The complex methanogenic environment is exposed to different environmental factors in the landfill. The methanogenic system together with the environmental factors has been studied in connection to landfills, anaerobic sediments, sludge fermentation technique etc. The most important factors are shown in Figure 2. In the

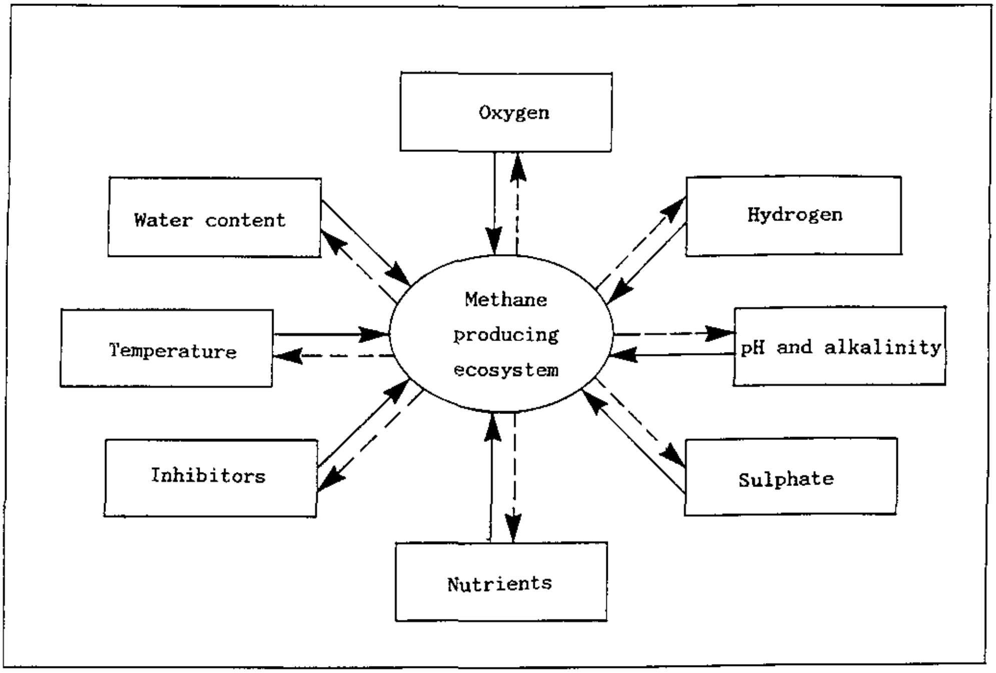

Figure 2: The most important environmental factors concerning the methane producing ecosystem.

following a brief comment to each of the environmental factors is given:

pH and alkalinity

pH is as mentioned a very important factor, since the methanogenic bacteria have a rather narrow optimum range (6-8.5), Zehnder et al. (1982). The production of fatty acids decreases the pH especially if the buffering capacity (the alkalinity) of the waste is low. A decrease of pH below 6 leads to very reduced methanogenic activity.

Sulphate

The methanogenic and the sulphate reducing bacteria competes for the same substates (acetate, hydrogen and methanol). Oremland & Polcin (1982) showed that the methane production was reduced when sulphate was present (in concentrations of 2000 ppm). Methane production *and sulphide reduction will usually proceed* simultaneously at landfills, due the inhomogeneity in waste composition.

Nutrients

Even though the waste in average not is lacking of nutrients (nitrogen and phosphorus) problems of nutrients deficiency is likely to occur in landfills due to the high degree of inhomogeneity in waste composition. For instance a bunch on newspapers disposed of the landfill is not easy degradable because of the very low nutrient content.

Inhibitors

High concentration of different compounds are able to inhibit the anaerobic degradation process. Concentrated solvents or oil waste leaking from a rusty drum into the surrounding waste, would lead to reduced degradation rates.

Temperature

The temperature of the waste is a very important factor. The optimum temperature for the methanogenic bacteria is 40-41°C (Hartz & Ham (1982) and Pfeffer (1979). The heat production form the anaerobic degradation is very low. It has however been shown by Rees (1980b) in big landfills (where the volume to surface area ratio is low), the low anaerobic heat production was sufficient to sustain a increased temperature in the waste (20-30°C raise in proportion to the temperature of the surroundings - see Figure 3).

Water content

Many investigations have shown that increased water content of the waste leads to increased methane production rates. (Buivid (1980), Rees & Grainger (1982)). Rees (1980a) found by comparison of different investigation, a logarithmic increase in gas production with increasing water content (from 0 to 60 percent water). The main reason for this is probably that the transfer of substates buffer, inhibitors and nutrients between different places in the landfill is higher at higher water content. In this way more optimal conditions are obtained in larger areas of the landfill.

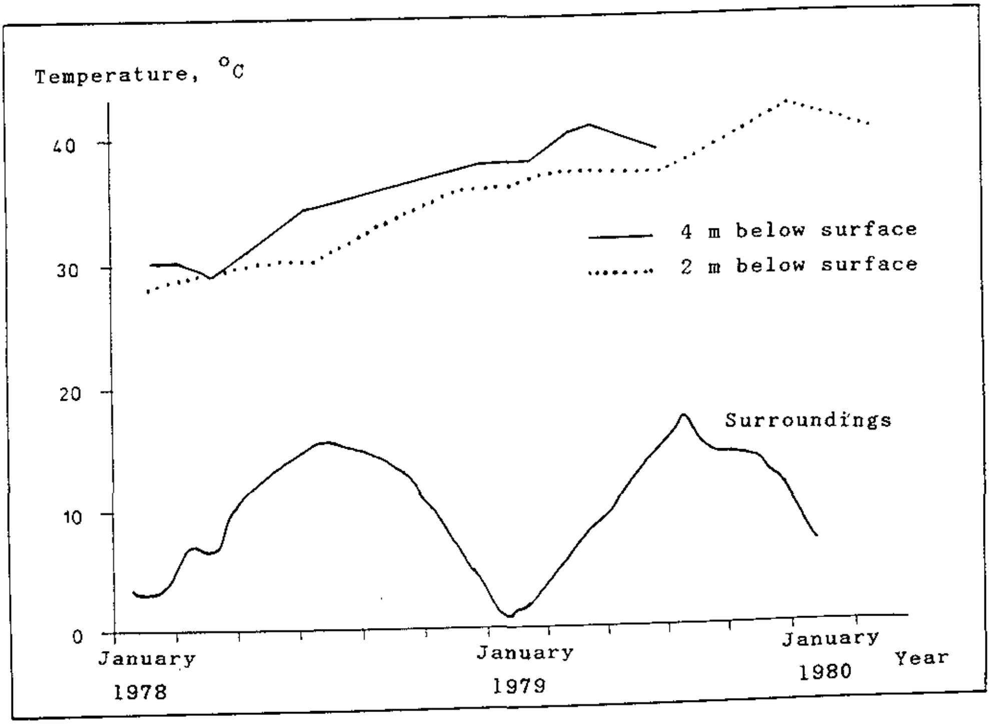

Figure 3: The temperature of the waste and the surroundings at a landfill site. Rees (1980b).

Oxygen

The presence of oxygen will totally stop any anaerobic activity.

Hydrogen

Hydrogen is produced by the fermentative and acetogenic bacteria (see Figure 1) and is used as a substrate by the methanogenic bacteria. High hydrogen content in the landfill gas is inhibitory to the degradation of fatty acids (propionic and butyric acids) McInerney, M.J. & Bryant, M.P. (1984). The hydrogen concentration is though a important factor in the symbiotic relationship between fermentalive and methanogenic bacteria.

Many of the environmental factors have been studies in laboratory scale. All the different factors make up a whole network of interdepending relationships. The transfer of this to a full scale landfill is obviously a difficult task.

ENHANCEMENT TECHNOLOGY

Many investigators especially in USA, England and West Germany have started on enhancing the degradation rates in landfills by technological operations help by the insight to the anaerobic system and environmental factors. The general scobe is to look at the landfill as a biological reactor.

Most investigations have been focusing on landfills in use, but some of the techniques can be used at abandoned landfills, and could serve as a remedial action to reduce environmental impact of the landfill.

Kjeldsen & Christensen (1987) reviewed the investigations of different enhancement technologies. In the following the most important conclusions of this review will be presented.

The enhancement technologies can be divided into technologies concerning the planning of sanitary landfills and the operation of the landfill.

Planning

Height of landfill

It is of advantage to construct sanitary landfills as high as possible. This is due to several facts:

- High landfills give low surface to volume ratios. The heat loss is therefore lower and a higher waste temperature can be established. The higher temperature gives higher degradation rates.
- A smaller landfill area is needed, which means that the generated leachate volume is smaller.
- The concentrations of organic matter in the leachate from high landfill are generally not higher than from shallow landfills since the lower layers of waste are working as a biological filter for leachate produced in the upper layers.

Stage size

Normally sanitary landfills are constructed in a number of stages. As small stage sizes as possible has its advantages:

- Final leachate recirculation can be established at an early stage
- Final gas collection systems (if desired) can be established in the early stage of the methanogenic phase.

Operation

Waste composition

The composition of the waste received at a landfill, is normally not controlled by what is favourable from a operational point of view, but more by the infrastructure of the surrounding area, the general waste disposal strategy, etc.

However a higher degree of control and registration of the recieved waste is desirable. The following waste types are to be handled separately:

- Clay soils is undesirable as daily soil cover (prevents flow of water through underlying waste layers). The soils must be kept for final soil cover.
- Calcareous wastes is to be kept in a temporary disposal site for additional buffering by mixing with other wastes
- Digested sewage sludges mixed into other wastes leads in general to a more optimal nutrient composition. The anaerobic degradation process is by the same time inoculated with methanogenic bacteria which often gives a shorter acid phase.

Pretreatment of waste

Especially in West Germany, waste has been pretreated by aerobic precomposting at the landfill site, Stegmann (1983). The first layer of waste is left uncompacted and uncovered. The most degradable compounds are degraded aerobically and a substantial increase in temperature is observed. After a couple of months the layer is compacted and covered with new waste. The aerobic removal of the most degradable compound reduces the risk for souring of the waste. Additionally the precomposted bottom layer serves as a filter layer for the leachate produced in the upper anaerobic layers of waste.

Mixing of received waste types

Thorough mixing of the received waste is a very important point. In this was you minimize the risk for local nitrient deficiency, souring due to a local concentration of highly degradable organic matter etc. A thorough mixing leads also to a more homogenous watering of the waste.

Recirculation of leachate

Recirculation of leachate is the most investigated technique of enhancing the degradation processes. Most investigations showed higher degradation rates by introduction of leachate recirculation, Stegmann & Spendlin (1985), Robinson & Marris (1985). The reasons for this is:

- The landfill serves as a biological filter for the recirculated leachate
- A higher transfer of substrates, nutrients etc. is obtained
- A higher water content is obtained

Recirculation of leachate can be carried out both at landfills in use and at abandoned landfills.

CONCLUSIONS

Landfilling of untreated mixed solid waste will still both in industrialized and developing countries be a common disposal technique. The understanding of the degradation processes in mixed landfills is therefore important and still much research is need especially by fullscale investigations.

The general picture is that the organic waste is degraded anaerobically to methane and carbon dioxide. Some of eventually codisposed chemical organic wastew are able to be codegraded in the anaerobic process while other chemical wastes (for instance oil waste) will leave the landfill unchanged.

A number of alternative landfill techniques have been shown to successfully enhance the degradation processes. The techniques are in particular usable at landfills in use, but the recirculation technique can also be employed at abandoned landfills. Most of the alternative techniques do not need advanced technology.

The effect of the employed techniques can however not be quantified, since landfills as biological reactors are all to complex and inhomogeneous. Early attempts to use mathematical models for predicting leachate qualities at fullscale landfills have not been successful.

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WATER AND ELEMENT TRANSFER FROM LANDFILL SITES

Dr. Ing. habil. H.-J. Ehrig

Summary

Municipal solid waste landfills represent an accumulation of material with less or more pollution potential. A part of this material is transfered to the environment by gas and leachate. The most important emission stream over a long time is leachte. A decrease of leachate quantity could not be estimated. This is only possible with additional top sealing systems. For many elements the period until concentrations below todays limiting values is some hundred years and more. The combination of decreasing concentration slopes with time and decreasing pollution limits could multiply the period of environmental risks.

INTRODUCTION

Municipal solid waste landfills are a mixture of different organic and anorganic compounds which represents an important but local limited pollution potential. Experiences during the last decades have shown that parts of this potential transfered to the environment by gas and leachate. Leachate as liquid emission is the most important risk for the environment. Measurements of quality and quantity at different landfills have shown that leachate is a complex and high polluted wastewater.

Considering the limited mass potential of each compound in a landfill it could be estimated that the transfer rate must be decrease versus time. Today the most serious problem of estimating the time of transfer rates above a given limit is the knowledge gap of biological, chemical and physical processes in landfills. Most of these processes could be described in general but not for the reality of a landfill. This paper should give some indications for water and element balances estimated from a lot of laboratory, pilot and full scale experiments.

WATER BALANCE

The water balances could be only observed at landfills which are sealed to surrounding strata. Fig. 1 presents leachate flow datas from different landfills as percentage of precipitation (operated and reclamated). With exception of 2 columns all values are lower than 25 % of precipitation (average 772 mm). If also two special cases with flow rates lower than 10 % excluded

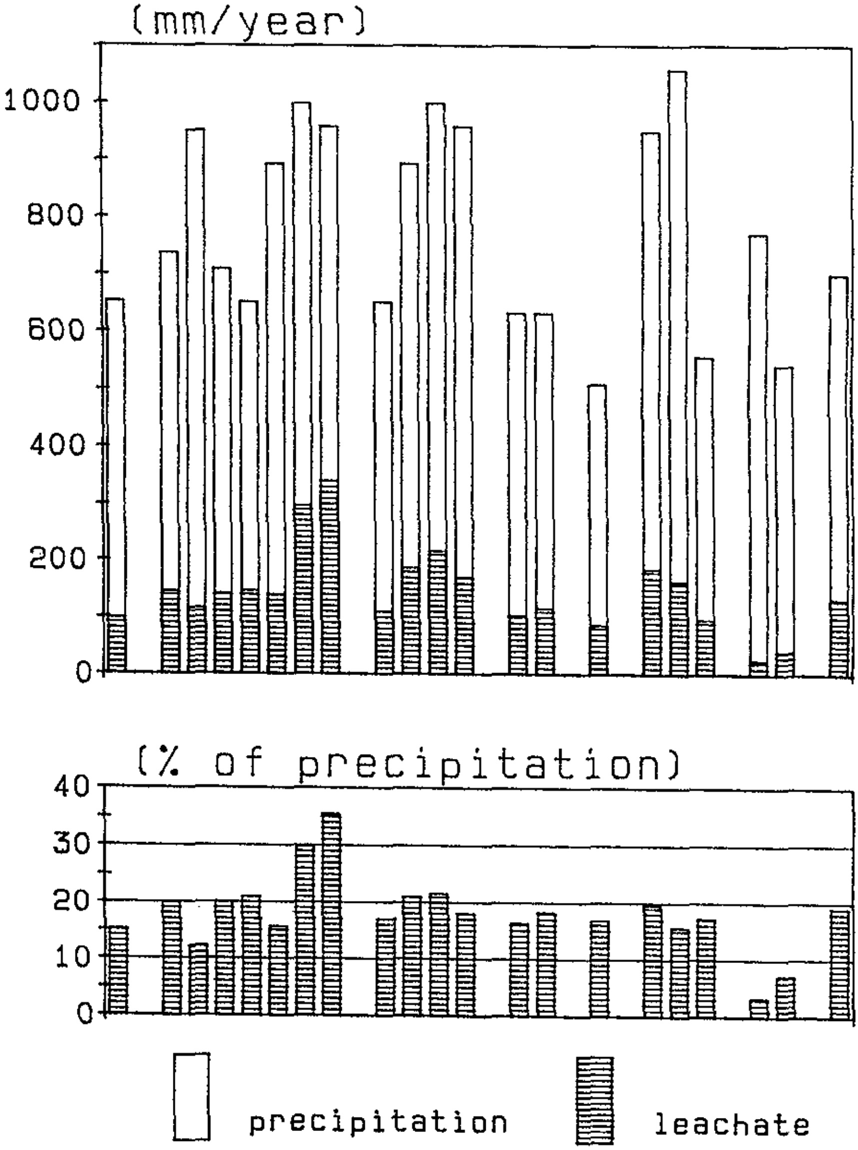

Fig. 1: precipitation (mm/year) and leachate flow (mm/year and % of precipitation) at different landfills and years

(very young landfills) the average percentage will be 17.9 %. For following calculations the value is rounded to 20 %. During first decades of a "landfill life" leachate flow is reduced by water consumption of anaerobic biological processes and increased water content of dumped refuse. Fill up of storage capacity and water consumption of biological processes in essential manner decrease both after some years or decades. As result the long termed leachate flow increases.

For both values > 25 % in Fig. 1 the end of storage capacity at this landfill could be calculated. The long time leachate flow must be estimated as precipitation minus evaporation. Under humic climatic conditions without top sealing 30 - 50 % of precipitation could be estimated as leachate.

LEACHATE COMPOSITION

With exception of carbon the most important transfer is the element flow by leachate. Therefore it is necessary to give some data of leachate composition and of influences on the concentrations.

During the past organics as BOD_5 or COD were the main leachate components. But now the significance of other parameter as nitrogen, metals and much more organic micropollutants increases sharply. An important fact is the change of organics (BOD_5, COD) with the change from the acetic to the methanogenic phase in landfills (see Fig. 2). The average trend of BOD_5 is marked with "1" for landfills build up in 2-m layers with 2-4 m per year. A build up of more than 4 m per year (marked with "2") increases the BOD_5 and COD-concentrations and delays the change of microbiological conditions in landfills. On the other hand a slower build up and other enhancements technics decreases the organic contents (marked with "3") with an earlier change to the methanogenic phase. Besides the absolute values of BOD_5 and COD the relationship BOD_5/COD is an important factor. Values greater than 0.4 during the acetic phase, indicates a good biodegradability. With the change to the methanogenic phase, the BOD_5 to COD-ratio decreases below 0,1, with low biodegradability of organic contents. High organic acid contents during acetic phase results in decreasing pH-values with increasing solubility of some anorganic substances (see table 1). The ammonium concentrations (Fig. 1) show only a low increase during the first years of operation and further relative constant values. An additional nitrogen component is organic nitrogen in the range of 30-150 % (average = 70 %) of ammonium.

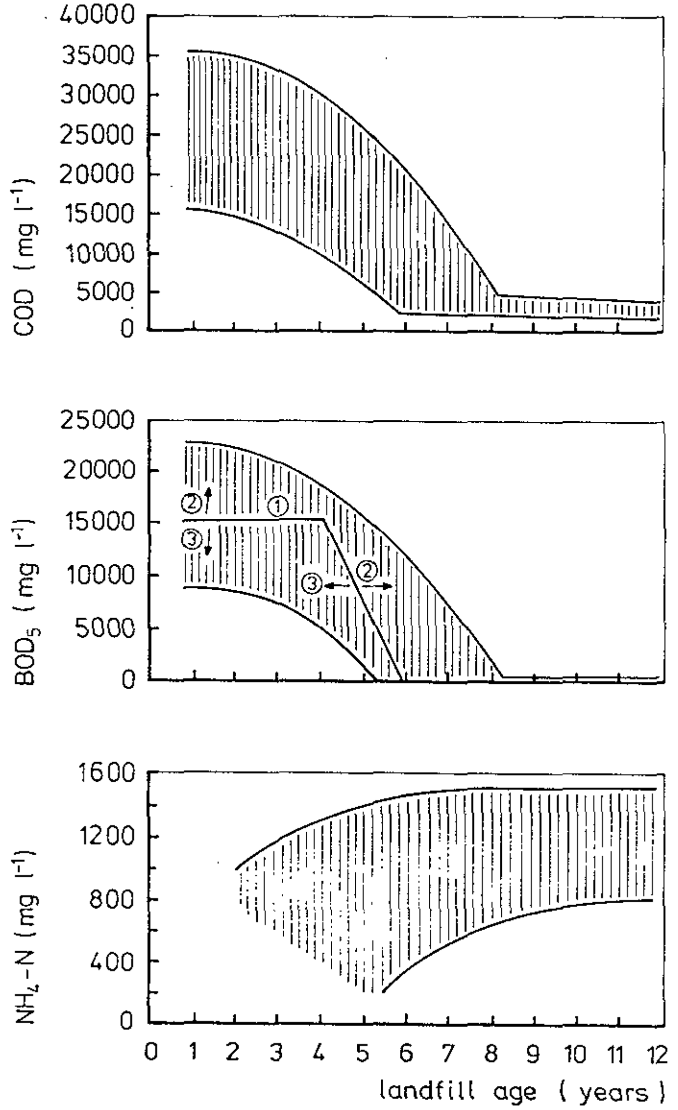

Fig. 2: general tendencies of COD, BOD_5 and NH_4 versus time

Table 1: Leachate analysis (parameters with differences between acetic and methanogenic phase)

	average	range
acetic phase		
pH (-)	6,1	4,5- 7,5
BOD_5 (mg/l)	13000	4000-40000
COD (mg/l)	22000	6000-60000
BOD_5/COD (-)	0,58	-
SO_4 (mg/l)	500	70- 1750
Ca (mg/l)	1200	10- 2500
Mg (mg/l)	470	50- 1150
Fe (mg/l)	780	20- 2100
Mn (mg/l)	25	0,3- 65
Zn (mg/l)	5	0,1- 120
methanogenic phase		
pH (-)	8	7,5- 9
BOD_5 (mg/l)	180	20- 550
COD (mg/l)	3000	500- 4500
BOD_5/COD (-)	0,06	-
SO_4 (mg/l)	80	10- 420
Ca (mg/l)	60	20- 600
Mg (mg/l)	180	40- 350
Fe (mg/l)	15	3- 280
Mn (mg/l)	0,7	0,03- 45
Zn (mg/l)	0,6	0,03- 4

Table 2: Leachate analysis (no differences between phases could be observed)

	average	range
Cl (mg/l)	2100	100- 5000
Na (mg/l)	1350	50- 4000
K (mg/l)	1100	10- 2500
alkalinity(mg $CaCO_3$/l)	6700	300-11500
NH_4 (mg N/l)	750	30- 3000
orgN (mg n/l)	600	10- 4250
total N (mgN/l)	1250	50- 5000
NO_3 (mg N/l)	3	0,1- 50
NO_2 (mg N/l)	0,5	0- 25
total P (mg P/l)	6	0,1- 30
AOX (μg Cl/l)*	2000	320- 3500
As (μg/l)	160	5- 1600
Cd (μg/l)	6	0,5- 140
Co (μg/l)	55	4- 950
Ni (μg/l)	200	20- 2050
Pb (μg/l)	90	8- 1020
Cr (μg/l)	300	30- 1600
Cu (μg/l)	80	4- 1400
Hg (μg/l)	10	0,2- 50

* adsorbable organic halogen

In table 1 the analysis divided into two groups. The first group represents values during acetic phase with high organic loadings and low pH-values. The second group shows values from methanogenic phase with low biodegradable organics and higher pH-values. Parameters in table 2 do not differ between phases. At all landfills with an age between 0 and 12 years no time slope occurs. The large variation of metal analysis is the result of differences between landfills and time depending randomised variation at each landfill. Until now differences between landfills cannot be explained by operation conditions etc.

MASS TRANSFER

Organics (Carbon)

The carbon transfer from landfills to the environment is the most complex transfer system:

- Organic carbon in refuse is not a homogeneous fraction. There are extremely differences between carbon in kitchen waste and greens on one side and in plastics or lignin on the other side. Until now decomposition rates of plastic and lignin also under landfill conditions are mostly unknown.
- Many of the organics in refuse are changed by biological, chemical or physical processes or by a combination of some or all. These processes are known in principle but the complex acting together is well unknown.
- Organic carbon in leachates is the result of biological processes. That is right for the short observed landfill age. But sure there are also other processes as solubility etc. which effects are coverd by the todays predominate biological processes.
- On the first view the carbon transfer by gas is a very simple way which could be easily calculated. But the whole gas production is only estimated today with differences between publisher of ± 50 %. All of these estimations are only used for a prediction of some decades. Statements for the long term biological processes are not possible.

Fig. 3 a presents COD data of operated landfills. Organic carbon was only measured in greater intervals. But it was possible to get a correlation between COD and TOC. It could be seen that after change to the methanogenic phase most values change to a more homogenous level of 3000 to 4000 mg COD/l. The level of methanogenic phase leachate shows also a slight time slope which could be extrapolated (Fig. 3b). From this and the gas flow calculation the main carbon transfer could be estimated over 100 to 150 years with very low TOC- or COD-values thereafter.

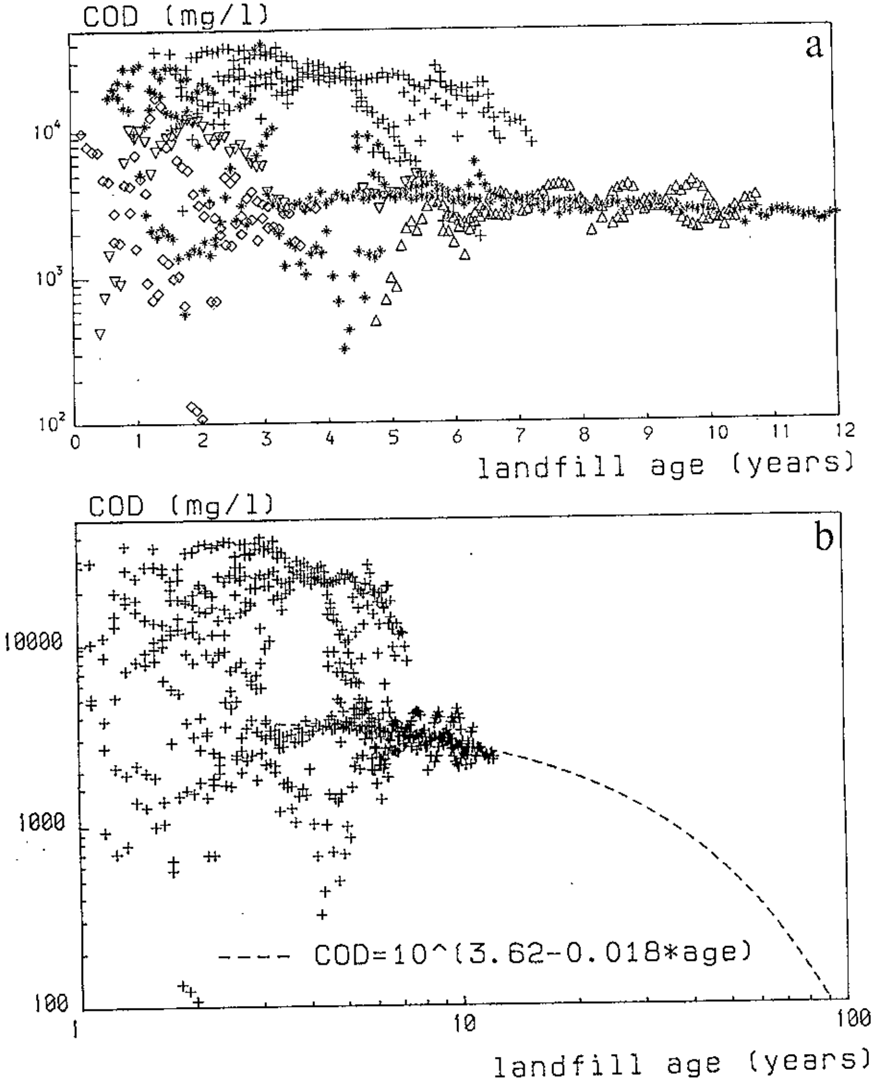

Fig. 3: a) measured COD-values of different landfill leachates
b) COD-extrapolation of landfill leachates (calculated from Fig. 3 a)

Choride

In opposite to carbon transfer the transfer of chlorides based mostly on chemical processes but influenced by biological processes. For the estimation of transfer rates two statements are necessary:

- Observations at landfills with a maximum age of 20 years show no decrease of chloride concentrations.
- The chloride potential of landfills as batch reactor is limited and therefore a decrease of concentrations must occur.

To close these differences we used the results of laboratory scale experiments (Fig. 4 a). But in opposite to real landfills the operation conditions are much more optimised. The connection point was the water regime with water contact per volume unit.

The relationship was 1 day lab. scale = 1,16 years full scale for solubility transfer (Fig. 4 and 5). The best fit of chloride concentrations was an exponential function (Fig. 4 a and b):

$C = C_0 \cdot e^{(-k*t)}$
t = time in years
k = coefficient = $\log_e 0.5/t_{0,5}$
$t_{0,5}$ = time of $C_0/2$
C_0 = initial concentration
= average chloride concentration of 2100 mg/l
C = concentration at time t.

The calculated values for $t_{0,5}$ were between 52 and 55 d (= 60 to 64 years). Fig. 4b and c presents the calculated concentration function and the sum of transfered chloride. The calculated transfer function only rises up to the lower part of the chloride potential area. Not transferable chlorides (e.g. in plastics) or organic chlorides (up to 25 %) could be an explanation for such differences. But also an increase of transferable potential as result of different biological, physical and chemical processes in future could not be excluded.

Nitrogen

The nitrogen transfer is a combination of carbon and chloride transport processes because most of nitrogen must be converted from organic nitrogen to ammonia by biological processes. As result the tendencies of ammonium concentration in Fig. 5 are much less exact as for chloride. Using the same calculation procedure as for chloride we have got half-life values of 75 to 82 years. Similar to chloride the calculated concentration values are shown in Fig. 5b and the transfer

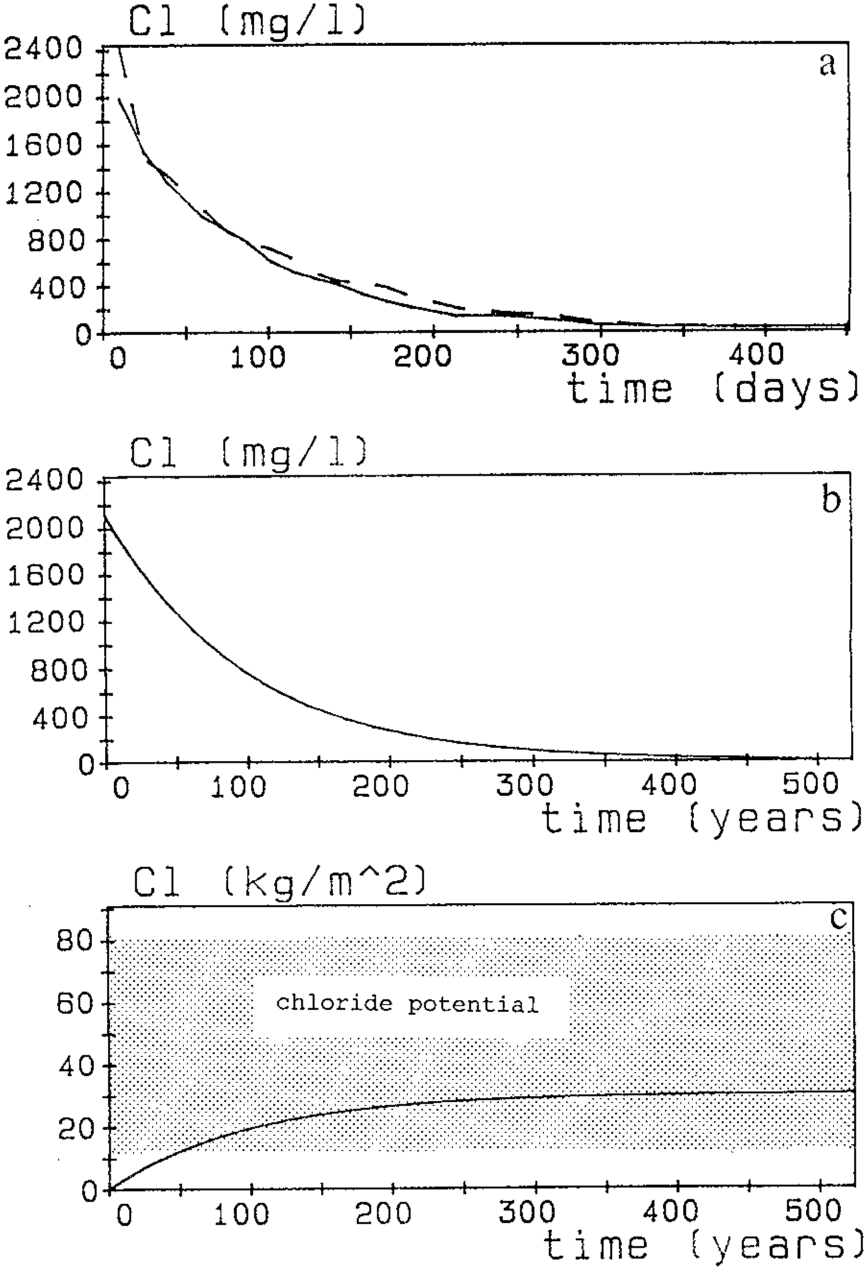

Fig. 4: a) measured chloride slope from lab. scale test cells
b) estimated Cl-concentrations of a full scale landfill (after Fig. 4 a)
c) chloride transfer (after Fig. 4 b)

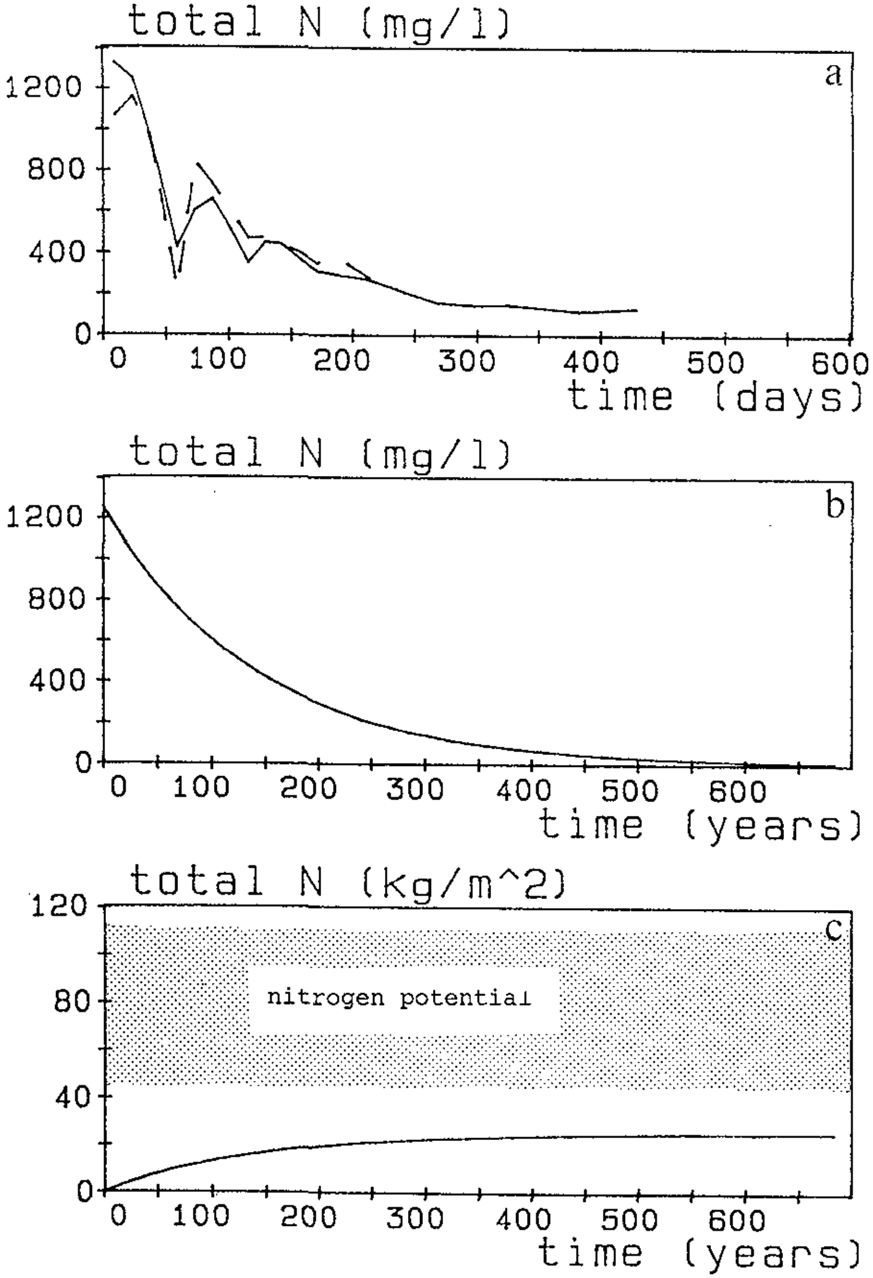

Fig. 5: a) measured nitrogen slope from lab. scale test cells
b) estimated N-concentrations of a full scale landfill (after Fig. 5 a)
c) nitrogen transfer (after Fig. 5 b)

function in Fig. 5c. Differences to measured nitrogen potential are much greater than for chloride. That could be a result of the much less uniform composition (chemical forms of nitrogen) and thereafter different solubility conditions of nitrogen in refuse. But similar to chloride discussion the knowledge gap is too large to explain such differences nearly exact.

Halogens and metals

The principle transfer functions must be nearly the same as for other compounds. But the relationship between measured initial concentrations and the potential in landfills is much higher. That means longer half-times must be estimated. As example:

potential 19,2 kg org Chloride (refuse column area = 1 m^2, high = 20 m); total org. Chloride transfer by gas = 0,053 kg org. Cl/m^2; org Chloride transfer by leachate = 0,03 kg Cl/$m^2 \cdot 100$ years.

Similar relationships are shown for some metals in table 3. For halogens and for metals very long emission time with concentrations near initial concentrations must be expected.

Table 3: Metal contents of refuse and transfer rates

parameter	contents mg/t dry refuse	transfer rate * mg/(t dry refuse*year)
Ni	≈ 15000	2,7
Cr	5000-100000	4,0
Cu	238000	1,1
Pb	399000	1,2
Zn	521000	8,0
Cd	3480	0,08
Hg	640	0,13

* refuse columnn : area = 1m^2 ; high = 20 m ; leachate production = 20% of precipitation

CONCLUSIONS

Without technical expedients leachate production must be estimated as an neverending process on a high flow level. The main leachate pollution parameters including metals are well known. But the knowledge of hazardous organic micropollutions in leachates and also in the solid waste is very low. With exception of carbon and thereafter BOD_5 and COD emission times could be estimated of at least some centuries. Emission times of organic and anorganic micropollutants would be much longer but cannot estimated nearly exact.

This paper should show that knowlegde of landfill processes etc. for technical purposes is sufficient but for all questions of the future interactions between landfills and environment there is a large gap. This deficiency cannot be closed because such long term processes cannot be shortened. All experiments with laboratory or pilot scale experiments could only give indications but not replace the reality. It is necessary to know more about true processes in full scale landfills. Experiences in the past often only engage values outside landfills - e.g. precipitation, leachate quality and quantity. gas composition - but not the interactions with processes in landfills.

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WATER BALANCE AT LANDFILLS

by Ole Stang, KEMP & LAURITZEN A/S

SUMMARY

The migration of groundwater pollutants from landfills is depending on the amount of leachate at the site. The amount of leachate is again depending on the precipitation and evaporation from the surface of the landfill.

Until now, several studies of Danish landfills have been carried out, and the outflows have been traced. The results of these studies show that the pollution intensity in the groundwater bodies below the disposal sites varies greatly from site to site. This is, of course, partly due to the chemical differences of the disposed materiel but it is also due to different groundwater recharge conditions.

In areas where groundwater recharge is large, pollution will rapidly occur in the groundwater zone. On the other hand there are landfills with hazardous waste were practically no trace of leachate can be found in the groundwater reservoirs due to small or absent groundwater recharge. In the latter cases the pollution is directed to surface waters or to the sea and only the upper groundwater bodies are contaminated.

In the paper the importance of the water balance for groundwater pollution is illustrated at a number of Danish disposal sites. From the presented data it is concluded that water balance considerations may be used together with geological information as means of preliminary evaluation of the pollution risk at a landfill. This goes also for preliminary evaluation of potential locations for new, safer landfills.

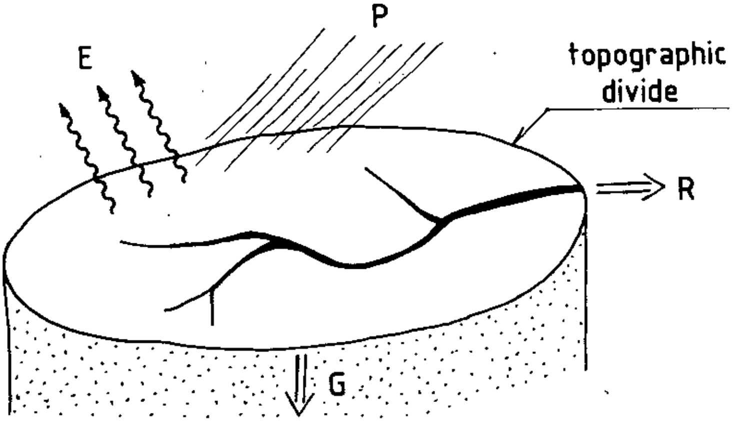

FIGURE 1 Water balance of a catchment area

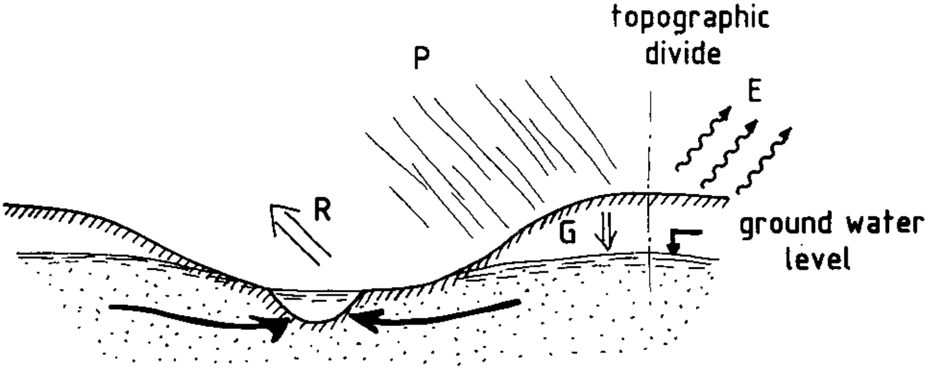

FIGURE 2 Cross section of a catchment area

1. INTRODUCTION

Evaluation of potential sites for new refuse dumps and investigation of existing dumps are complex tasks. The siting of new dumps should be as "safe" as possible, i.e. the risk of pollution of groundwater and surface water bodies should be minimized. Therefore new dumps are equipped with lining in the bottom and covered with lining in order to prevent the formation of leachate. Usually, the geological conditions in the area are also studied in detail in order to reveal wether the water bearing formations are protected by aquicludes of clay or other low permeable soils.

Another important aspect that should be considered is the water balance of the area. By analyses of the distribution of the pollution from existing refuse dumps it is possible to disclose the importance of the water balance.

2. WATER BALANCES

The study of the water balance for a specific land area is often applied by hydrologists, when river run-off is measured or calculated. Also groundwater modelling involves the water balance of an aquifer.

The traditionel use of the water balance concept is related to calculation of river run-off from precipitation. If the water balance is set up for a specific catchment area and a period of several seasons it may be formulated:

$$R = P - E - G$$

where R is the river run-off, P is the precipitation, E the evapotranspiration and G is the net groundwater recharge in the catchment or the net outflow of groundwater from the catchment area. (See fig. 1) It is assumed that the changes of water content of the reservoirs in the area are negligible. An equation like the above stated one may, however, cover big differences from one part of the catchment area to another. This is shown in figure 2. The figure illustrates, that the position of the groundwater level is very important for the local formation of groundwater.

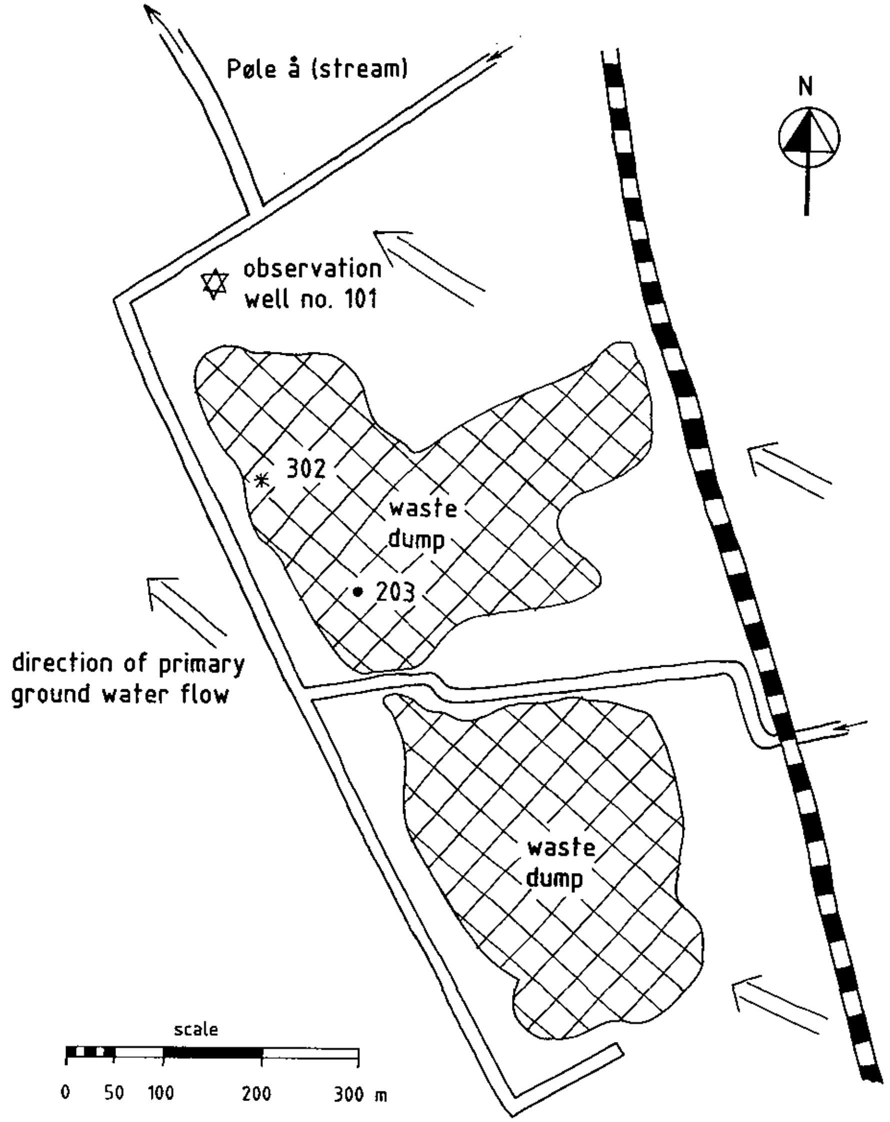

FIGURE 3 Hillerød refuse dump

Groundwater recharge mainly occurs in areas near the topographic divides and far from streams and rivers. In the flat marsh areas near the rivers the groundwater level is often so near the surface, that precipitation is transformed into surface or near surface run-off.

This is the reason, why refuse dumps located in valleys near streams may be considered to be less critical with regard to groundwater contamination. This also goes for locations near the coast. But the pollution risk with regard to the nearby stream or sea is, of course, larger at these locations than it is near the topographic divides.

The above remarks and figures are simplified and only the situation with one groundwater reservoir is considered. In real life geological differences also play a major role in the assesment of "safe" locations. Another very important factor, not yet mentioned, is the location of well fields. If well fields are located near the rivers or the coast, this should drastically influence the choise of location for new refuse dumps.

3. EXAMPLES OF POLLUTION FROM WASTE DISPOSAL SITES

3.1 Hillerød refuse dump

Since 1959 refuse has been dumped at a 0,35 squarekilometer area south of the town of Hillerød. The dump site is located at the upper reaches of the stream "Pøle å" in a former bog. Approx. one kilometer northwest of the dump site a well field is located. The yearly abstraction from this well field is some 2 mill. m^3.

At this background hydrogeological and chemical investigations have been carried out in the period 1985-87. These investigations have comprised the drilling of 24 unscreened wells, 8 wells to upper secondary groundwater reservoirs, 5 wells to lower secondary reservoirs and 3 wells to the primary reservoir. Water samples from these wells have been analysed for a wide range of chemical parameters.

	Upper secondary aquifer	Lower secondary aquifer	Primary aquifer
Well no.	202	302	101
Depth (m)	9	22	51
Ammonium-N (mg/l)	403	0,83	0,62
Chloride (mg/l)	424	42	31
Sodium (mg/l)	360	22	23
Potassium (mg/l)	318	3,7	3,2
Phenols (µg/l)	25	not identified	not identified

TABLE 1 Chemical analyses of groundwater at Hillerød refuse dump

In table 1 a few of these results are listed for three wells screened in different levels. It should be noted that water sampling have not taken place at the same time for the three wells.

From the table it is evident that the leachate from the dump site hardly can be traced in 22 meters depth or 51 meters depth. As the secondary reservoirs are sandlenses in boulder clay it is not likely that the horizontal water movements are of great importance in the sense that they could account for the dilution of leachate. The dilution process could, however, play a major role in the primary chalk aquifer.

The water balance for the catchment area of the upper "Pøle å" has also been evaluated. Rainfall data from a nearby meteorological station and evaporation data from a climatological station have been used to calculate the net precipitation. A river gauge has been established in order to measure run-off from the 2,1 km^2 catchment area around the dump site. The water balance for the hydrological year 1986-87 shows that the groundwater recharge is negligible in the catchment area where the dump site is located.

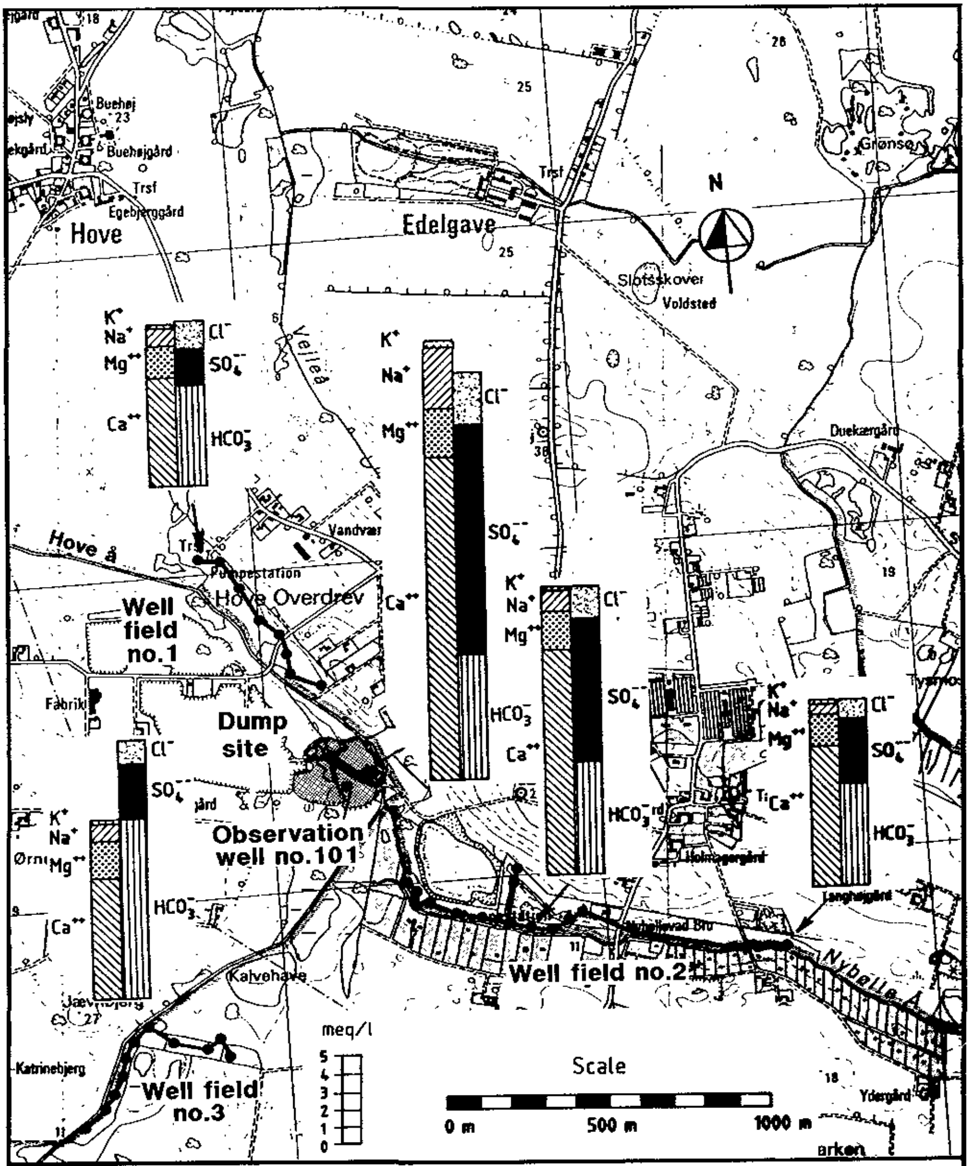

FIGURE 4 Hove refuse dump

Both the chemical analyses and the water balance considerations then supports the assumption that the risk of a future contamination of the primary aquifer is small, provided that the head of the primary aquifer is not lowered by increased groundwater abstraction. The investigations implies that most of the contamination from the dump site is received by the surface water bodies in the area. This is also confirmed by chemical analyses carried out on water sampels from the stream.

3.2 Hove refuse dump

Preliminar investigations have been carried out at Hove refuse dump close to "Hove å" (figure 4). The dump site is located in an area where intensive groundwater abstraction takes place. As shown in figure 4 three major well fields are situated within a radius of one kilometer from the dump site.
A number of small wells have been drilled in the dump area in order to determine the extent of the deposits and the composition of the leachate. The mean concentration of a few, selected chemical parameters in the leachate is shown in Table 2.

	Leachate	Primary aquifer	Primary aquifer
Well no.	mean of four wells	101	mean of five analyses on well-field no. 3
Depth (m)	5-10	25	25-50
Ammonium-N (mg/l)	16	0,88	0,18
Total - P (mg/l)	0,56	0,066	0,013
Chloride (mg/l)	536	94	46
Sodium (mg/l)	404	75	22
Potassium (mg/l)	100	9,8	2,7

TABLE 2 Chemical analyses of groundwater at Hove refuse dump

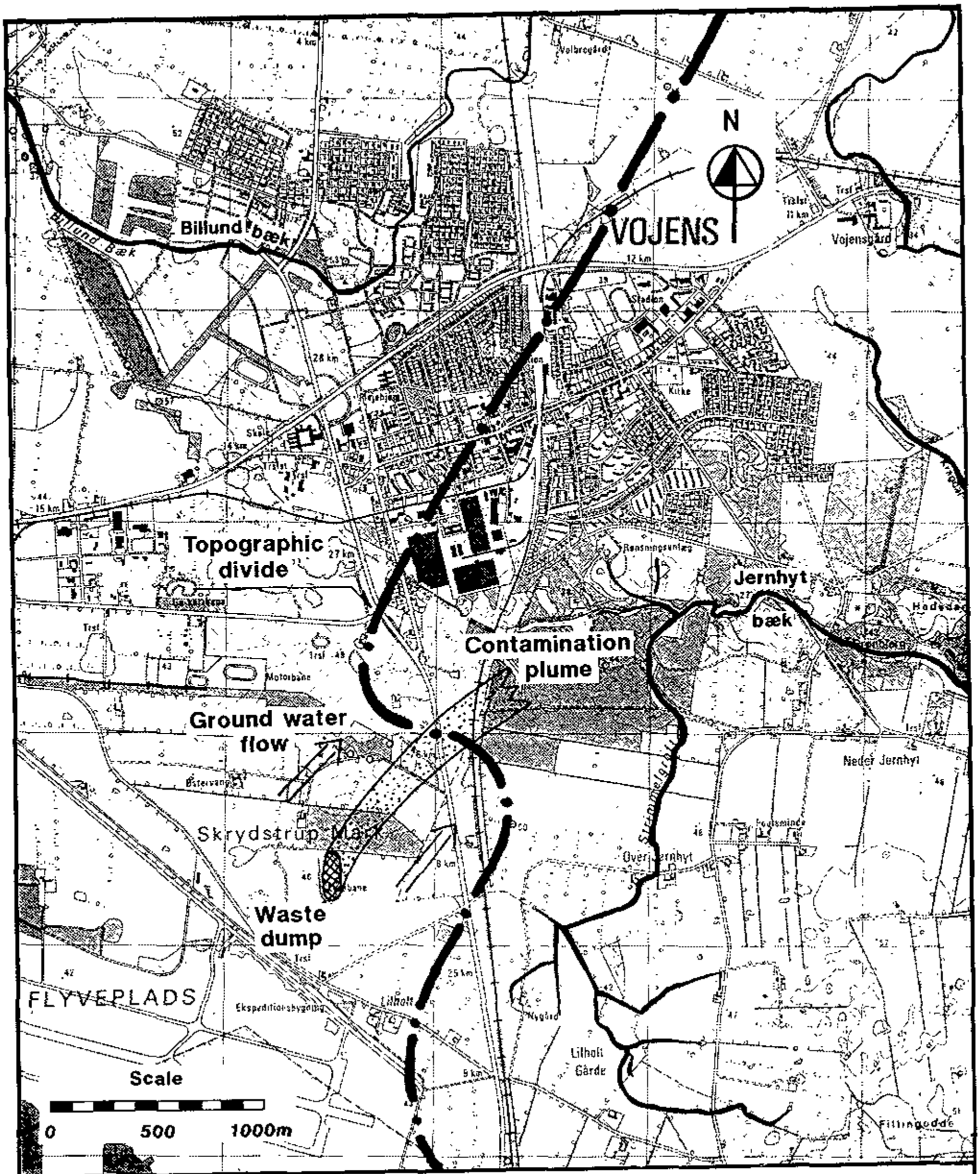

FIGURE 5 Skrydstrup refuse dump

A deep observation well (no. 101) has also been drilled at the landfill. The chemical composition of the groundwater in this well represents the primary chalk aquifer. Water analyses from well field no. 3 is assumed to represent groundwater that is not contaminated from the dump site, as it is situated in the upstream direction. In figure 4 is also shown the ionic strength of the water sample from well no. 101 and an average ionic strength for the three well fields in the area.

In this case it is evident that the primary aquifer underneath the dump site is affected by the refuse deposite, although the dump site is situated in a low marsh area near a stream. The probable reason for this is that the well fields near the dump site over the years have caused considerable drawdowns of the hydraulic head. This has induced leakage from the dump site.

3.3 Skrydstrup refuse dump

The location of the refuse dump is shown in figure 5. In this case the dump site is situated only half a kilometer from the main topographic divide of Jutland. Furthermore the soil is sandy and the groundwater table is around 8 meters below the deposits. This means that groundwater recharge is large and the aquifer vulnerable. A large portion of the deposits at the dump site have contained chlorinated organic solvents and the soil in the deposits have been heavily contaminated with these solvents.

The solvents can also be traced in the groundwater at concentrations up to 8000 µg/l close to the dump site, and at a distance of about 1 km in the downstream direction the concentration is about 40 µg/l. Ultimately the stream "Jernhyt bæk" may be threatened by the pollution, as part of the groundwater flow emerges in this stream. Also a waterworks about 2,5 kilometers downstream of the dump site is threatened. The case is, however, a very good example of a waste dump, where the groundwater is receiving the major part of the pollution.

Remedial actions have been employed at the dump site. Several tons of heavily polluted soil and chemicals were excavated and transported to destruction and the less polluted soil is moved to a controlled deposit site near the original location. Also a project for cleaning up the groundwater pollution has recently been launched.

4. CONCLUDING REMARKS

From the three cases mentioned above it may be concluded, that the groundwater recharge conditions play a major role, when the risk of pollution from a landfill shall be assessed. The groundwater recharge is depending on the natural water balance of the area and the local hydrogeological conditions. The introduction of heavy groundwater abstraction will, however, disturb the natural water balance and induce increased recharge.

If the risk of groundwater contamination shall be minimized, the dump sites should preferably be located in low marsh areas near the coast or near streams.

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ASSESSMENT OF LONG-TERM IMPACT OF METAL-RICH WASTE MATERIALS ON GROUNDWATER

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Summary

Test procedures are presented for evaluation of long-term release of metals from waste materials into groundwater. (1) Suitable solid substrates in dialysis bags have been deposited for a certain period of time into groundwater aquifers. With the in-situ method individual critical pollutants as well as different receiving media and depositional milieus can be compared under realistic complex conditions. (2) Using a laboratory circulating system including ion-exchanger unit the "quick motion" effect is attained by controlled significative intensivation of the relevant parameters pH-value, redox potential and temperature. In both procedures, by means of a sequential extraction of the contaminated solids before and after exposure, a qualitative and quantitative assessment of mobilisation-relevant "pools" can be reached, by which the release can be predicted beyond the experimentally simulated time.

INTRODUCTION

Following the detection of contaminated sites the question is usually raised by what remedial measures future deleterious impact particularly on groundwater can be avoided. Similarly, for approval of new depositions of contaminated materials, authorities have to impose measures which "sufficient safely exclude adverse long-term effects". In both cases relevant criteria are lacking, particularly with regard for prognosis of the long-term behaviour of pollutants. The aspect of long-term effects will increasingly be included in the management of waste disposal sites. While in the past waste disposal performance was more or less evaluated in terms of short-term costs, there is now a re-orientation in the direction of taking into account the long-term consequences - i.e., for decades and centuries to come - of waste disposal methods. In this light, particular attention is given to the deposition of residues whose utilization does not appear meaningful from an ecological point of view, or would require disproportionate resources (Böhlen, Baccini et al., 1988). Such materials would primarily include waste incineration ashes. One of the major questions in this new conceptual perspective will be to what extent these materials are fulfilling the requirements of a "final storage quality", i.e. the similarity with natural processes and materials. The present work describes a new method for evaluating the long-term behavior of trace elements in inorganic waste residues.

APPROACHES TO PREDICTION OF LONG-TERM BEHAVIOR OF METALS IN GROUNDWATER AQUIFERS

The nature of the transport system or "solvent vehicle" (Fuller & Warrick, 1985a) carrying the waste components requires that certain parameters be incorporated in a test procedure, if it is to be representative of the real environment. This in particular pertains to the characterization of reactivity and mobility of pollutants. Various laboratory techniques have been reported for generating leachate from hazardous waste and are generally grouped into batch and column extraction methods. The *batch extraction method* offers advantages through its greater reproducibility and simplistic design, while the *column method* is more realistic in simulating leaching processes which occur under field conditions. It has been noted by Jackson et al. (1984) that relative standard deviation of experimental data was over twice as great for the column method than for the batch method; this result was attributed to varying degrees of "channeling" that may have occurred during leaching of waste constituents using the column method. The soil-column technique requires that only the readily measurable properties of the disposal system be determined. These include (1) soils data (physical and chemical), (2) transport fluid data (e.g. total organic carbon, soluble common salts, pH, etc.), (3) soil-column data, and (4) breakthrough curves and statistical analyses (Fuller & Warrick, 1985b).

Initial estimation of potential release of metal from contaminated solids is mostly based on *elutriate tests*, which - apart from the characterization of the reactivity of specific metals - can provide information on the behavior of metal pollutants under typical environmental conditions. Common single reagent leachate tests, e.g. U.S. EPA, ASTM, IAEA, ICES, and DIN use either distilled water or acetic acid (Theis & Padgett, 1983). A large number of test procedures have been designed particularly for soil studies; these partly used organic chelators such as EDTA and DTPA, both as single extractants or in sequential procedures (Sauerbeck and Styperek, 1985). Physiological solutions have been applied, e.g. for the estimation of organ-specific effects of atmospheric particulates (Harris and Silberman, 1983, 1988). Best results with respect to the estimation of short-term effects can be attained by "cascade" test procedures at variable solid/solution ratios: A procedure of the U.S. EPA (Ham et al., 1979) designed for studies on the leachability of waste products consists of a mixture of sodium acetate, acetic acid, glycine, pyrogallol, and iron sulfate.

With respect to the effect of individual influencing parameters, lowering of pH-values, either from acid precipitation or from oxidation of sulfidic minerals, seems to be particularly efficient play a dominant role in the mobilization of trace elements from soils, sediments and solid waste materials. A "mobility test" procedure for soils based on *variations of pH-values* has been proposed by Kiekens & Cottenie (1985). Application to a large number of polluted and non-polluted soils indicate that typical mobilisation patterns are obtained for different heavy metals. Beside the nature of the element, the pH-curves reveal typical textural features of different soil substrates. For the study of *combustion residues* a standard leaching test has been developed by the Netherland Energy Research Centre (Van der Sloot et al., 1984). In the column test a column is filled with the material under investigation and percolated by acidulated demineralized water (pH = 4; for evaluating most relevant effects of acid precipitation) to assess short- and medium-term leaching (< 50 years). In the cascade test the same quantity of material is extracted several times with fresh demineralized water (pH = 4) to get an impression of long-term leaching behavior (50-500 years). As a time scale the liquid/solid ratio (L/S) is used; the relation between this relative time scale and the actual time scale depends on the time required for a L/S ratio to be reached in the actual situation. The maximum leachability is assessed by a shaking experiment at L/S ratio of 100 under mild acid conditions (De Groot et al., 1987). Recent improvements of this method have been achieved by comparing the "L/S-curves" for an individual element with its stability in a wider pH-spectrum; in some cases direct mineralogical evidence can be given for distinct metal compounds (Van der Sloot, private communication).

In connection with the problems arising from the disposal of solid wates, particularly of dredged materials, *chemical extraction sequences* have been applied which are designed to differentiate between the exchangeable, carbonatic, reducible (hydrous Fe/Mn oxides), oxidizable (sulfides and organic phases) and residual fractions. The undisputed advantage of this approach with respect to the estimation of long-term effects on metal mobilities lies in the fact, that rearrangements of specific solid "phases" can be evaluated prior to the actual remobilisation of certain proportions of an element into the dissolved phase (Förstner, 1985).

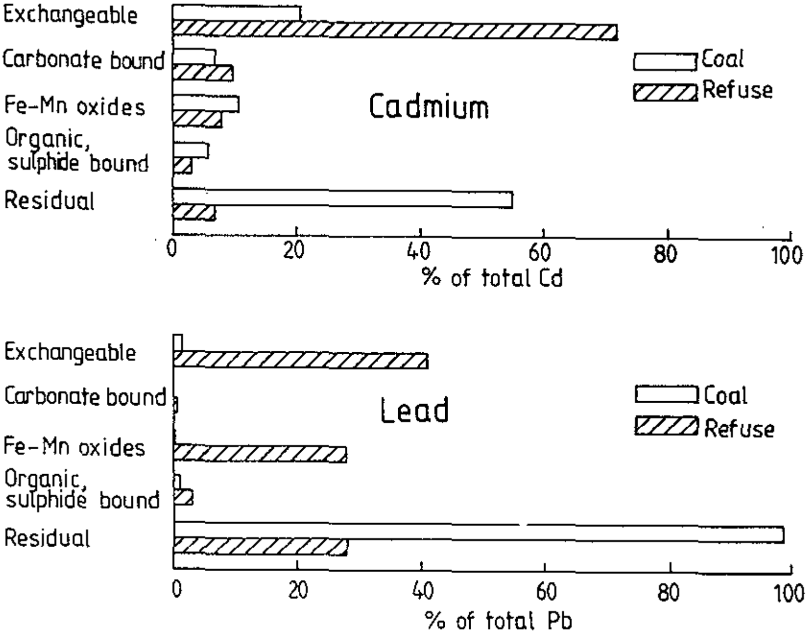

Figure 1: The Chemical Associations of Cadmium and Lead in Coal Fly Ash and Refuse Fly Ash (Wadge and Hutton, 1987). Extraction Sequence after Tessier et al. (1979)

Waste incineration ashes usually exhibit relative high concentrations of trace metals - *Zn and Pb* up to the percent range - and particularly strong enrichment factors compared to natural contents have been observed for these elements and for cadmium and silver (Brunner and Zobrist, 1983). Sequential extractions performed by Wadge & Hutton (1987) indicate (Figure 1), that about 20% of total Cd and 1% of total Pb in coal fly ash was in exchangeable fraction; in contrast, the single largest fractions of Cd and Pb in refuse ash, at 72% and 41%, respectively, were present in the exchangeable form. It is suggested that this effect is mainly due to higher concentrations of metal associations with chlorides (Baccini & Brunner, 1985). Measures before incineration include reduction of PVC in the waste; after incineration washing of the residues can be performed either with neutral or acidified water (Hämmerli-Wirth, 1987).

IN-SITU-EXPERIMENTS ON CHANGES OF SOLID HEAVY METAL PHASES IN AEROBIC AND ANAEROBIC AQUIFERS

Predictive models are limited due to the difficulties to describe sorption kinetics between (persistent) chemicals and solid phases. In such cases it could be advantageous for directly measuring changes of chemical forms of typical soil and waste constituents at in-situ conditions with respect to interstitial water composition, e.g. by inserting dialysis bags containing typical substrates with varying metal dosages were inserted into boreholes. Fig. 2 shows initial results on the behaviour of low metal-dosage to iron oxyhydrates and solid organic substrates under anaerobic conditions. In both examples lead is released during the 4-weeks experiment particularly from easily reducible phases. For copper slight increase is observed during the experimental period, again in the easily reducible fraction (Förstner & Carstens, 1988).

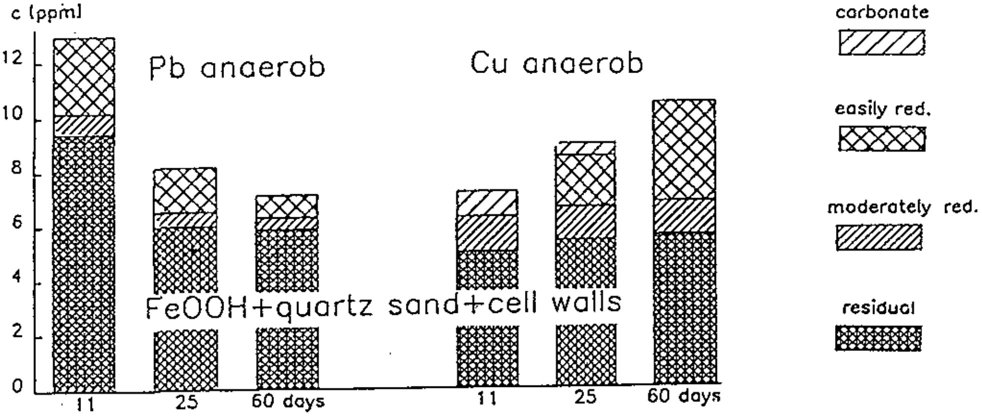

Figure 2: Changes of Total Concentration and Chemical Forms of Pb and Cu on Inorganic and Organic Substrates Inserted into Anaerobic Groundwater (Förstner & Carstens, 1988)

With the *direct method* relationships between anthropogenic substances and waste/natural soil substrates can be evaluated for: (i) site-specific properties of different disposal systems, which can be compared with respect to their compatibility with the waste material, consisting of a wide range of pollutants; and (ii) influences of a stabilized system of hydrological and chemical factors, mechanisms and processes - including biochemical interactions - on the behaviour of individual critical pollutants.

DETERMINATION OF RELEASE RATES OF HEAVY METALS FROM SOLID MATTER BY "QUICK-MOTION" EXPERIMENTS

The procedures described so far are restricted with regard to prediction of long-term effects in waste deposits, since these concepts neither involve mechanistic nor kinetic considerations and therefore do not allow calculations of release-periods. This lack can be avoided by controlled *significative intensivation of the relevant parameters* pH-value, redox potential and temperature combined with an extrapolation on the potentially mobilizable "pools", which are estimated from sequential chemical extraction before and after treatment of the waste material. In addition, there is the question on "realistic assumptions" regarding the hydrological conditions within and around the deposit. If permeability is low within the deposit due to dense packing of the materials compared to the surrounding groundwater aquifers one should expect flow-by conditions of the surface water rather than a flow through the deposit. On the other hand, extreme conditions may occur during erosion of the deposited waste substances, affecting intensive interactions between water and critical components. The design of the kinetic experiments for evaluating possible long-term effects has to include all these possibilities.

Experimental Procedure

Long-term behaviour of metal contaminants (including aresenic, cadmiuim, thallium, and vanadium) in *metal oxide residues from heat processing*, which were deposited into borrow pits from sand removal, has been studied at a 225 hectare site in the lower Rhine valley near Düsseldorf (Schoer & Förstner, *1988*). An experimental scheme, which has originally been used by Patrick et al. (1973)

and Herms & Brümmer (1978) for the study of soil suspensions and municipal waste materials, was modified by inclusion of an ion-exchanger system for extracting the metals released within one week time each (Figure 3). Solutions adjusted - partly self-regulating - to combinations of pH 5/8 and redox 0/400 mV circulate with 2 litres per day through columns, containing 1:4 mixture of solid waste with quartz sand, the latter component to improve permeability.

For most elements studied acidity is more effective than changes in redox conditions. Remobilized element percentages in a typical sample were approx. 20% to 40% of total concentration for Zn, Co and Cd at pH 5 (both 0 mV and 400 mV); higher release of Pb (29%) and Cu (8%) was found at +400 mV than at 0 mV (both 2%). Maximum mobilization of thallium occurred at pH 8 and 400 mV (25%). Solubility of Cr was less than 1% for all studied conditions.

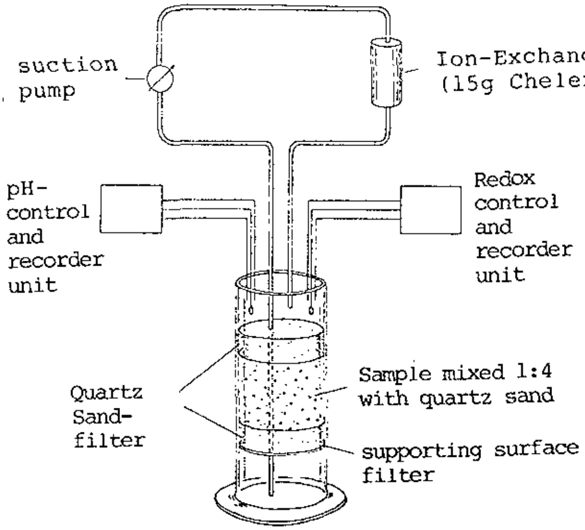

Figure 3: Experimental Design for Long-Term Prognosis of Metal Release (Schoer and Förstner, 1987)

Release Kinetics

Temporal release patterns are different for the individual elements (Figure 4): While at pH 5/400 mV release of cadmium seems to be completed within the experimental period mobilization of copper is still going on and the end point cannot be estimated from the data of the "kinetic" experiments. The same effect has been found for the examples of thallium and vanadium. For the other elements, the endpoint of release can be determined as approximately 10 mg cobalt, 0.6 mg cadmium, 600 mg zinc and 0.3 mg chromium, 2 mg barium, and 20 mg lead (per 100 g of solid substrate treated with 140 L solution).

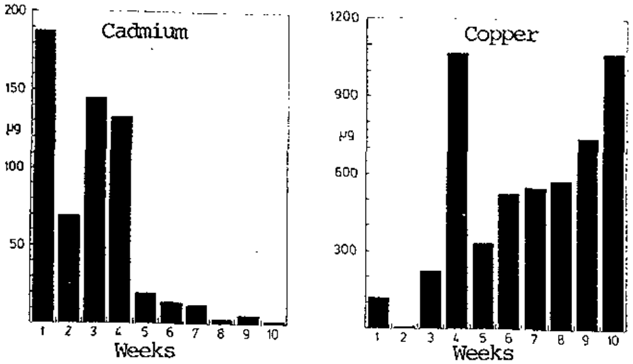

Figure 4: Temporal Evolution of Cadmium and Copper Release (in g). 100 g Sample with 140 L Solution at pH 5/+ 400 mV

Experiments on flow-by conditions have been performed with a hematite-rich sample over a period of 4 weeks at pH 5/400 mV conditions at two temperatures (10° and 30°C). In Table 1 a comparison is made of the maximum release effects in x-th week for flow-through (different substrates) and flow-by experiments (10°C values). In particular the data of elements, which have been studied under the same experimental conditions - pH 5/-+400 mV - clearly indicate the much reduced mobilization of metals in the flow-by experiments compared to the flow-through study. For cadmium, zinc, and lead, maximum release during flow-by conditions is less than 1 percent of the flow-through conditions.

Table 1: Maximum release of elements from 100 g sample in x-th week under flow-through (different experimental conditions on various substrates) and flow-by conditions (hematite-rich sample, pH 5/400 mV, 10°C)

Element	Flow-through experiments Max.conc.	Condition	x-th Week	Flow-by experiments Max.conc.	x-th week
Zinc	180.000 µg	(Qu-5/400)	1st week	1.000 µg	1.-4.week
Lead	5.000 µg	(He-5/400)	4th week	40 µg	1st week
Cobalt	4.000 µg	(Qu-5/400)	1st week	150 µg	3rd week
Vanadium	>2.000 µg	(Mg-5/0)	>10th week	<1 µg	1.-4.week
Copper	>1.000 µg	(Qu-5/400)	>10th week	10 µg	4th week
Arsenic	1.000 µg	(He-5/0)	6th week	2 µg	1.-4.week
Thallium	400 µg	(He-5/0)	7th week	40 µg	3rd week
Cadmium	200 µg	(Qu-5/400)	1st week	<0.5 µg	1.-4.week
Chromium	70 µg	(He-5/400)	3rd week	<2 µg	1.-4.week

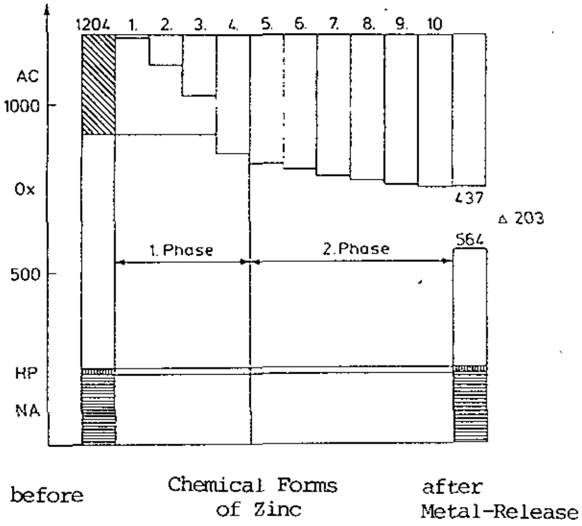

Figure 5

Comparison of zinc-"pools" in a sample of waste material before and after treatment by pH 5/400 mV solutions.

AC = NH_4-Acetate
OX = Oxalate-Buffer
HP = H_2O_2-Extract
NA = Residual-Fr.

Extrapolation from Mobilizable Metal "Pools"

In particular for the elements, of which the endpoint of release cannot be estimated from the respective cumulative curves of the "kinetic experiments", extrapolations with respect to total mobilizable metal "pools" can be performed by comparing data from sequential extraction experiments. Taking the example of zinc in Figure 5 the more labile "exchangeable" fractions should be released at first ("phase 1"), whereas during "release phase 2" - which is much slower than initial mobilization of acetate-extractable zinc - part of oxalate-reducible compounds are dissolved.

Taking into account both element contents released during the 10 week experiments and those extrapolated from reducible pools concentrations can be calculated for the interaction of 100 g solid waste with 140 litre of pH 5/ 400 mV (Tl: pH 8/ 400 mV) solution (*Table 2*). *At these extreme* assumptions with respect to both solute contact and interactive mechanisms most metal concentrations would be expected in the order of magnitude of guideline values set by the state authorities of Nordrhein-Westfalen for aqueous elutriates of waste materials class 1 (partly derived from drinking water standards). Further considerations with respect to more realistic hydrological conditions should be focussed on the examples of zinc and lead, where excessive concentrations could be derived from the present model (Table 2).

Table 2 Calculation of Dissolved Metal Concentrations in Landfill Leachates *from Direct Determinations* (Treatment of 100 g Sample with 140 Litre Extraction Solution) and from Estimations of Mobilizable Metal Pools.

Mobilization at Flow-Through Conditions*			Mobilization-Relevant Metal-"Pool" in mg/kg				Results in Diss. Concentr.	Depot Class I Criteria
	per kg	pH/Redox	Acetate (1)	Oxalate (2a)	(2b)	H_2O_2 (3)	mg/L	mg/L**
Cd	6 mg	pH 5/ 400	[1		2	1]	0.004	0.005
Co	100 mg	pH 5/ 400	[6	90]	40	5	0.07-0.1	0.05
Cr	3 mg	pH 5/ 400	0		250	15	0.002	0.01[a]
Cu	50 mg	pH 5/ 400	[15	35]	185	50	0.03-0.20	0.10
Pb	600 mg	pH 5/ 400	[20		350	200]	0.40	0.05
Tl	20 mg	pH 8/ 400	[10	10]	30	5	0.01-0.03	0.01
Zn	6000 mg	pH 5/ 400	[1000		5000]	300	3.0	1.0

*140 L solution/100g solid waste **Nordrhein-Westfalen (Anon.'82)

[a] value for Cr(VI)

Differences in the temporal development of release rates for the individual elements are connected with their sorption/desorption behaviour, which is primarily due to pH-effects but may also be influenced by complexation, e.g. by elevated concentrations of chloride ions. With respect to the pH-effects, however, there are significant differences in the response of the various solid substrates to the addition of H^+-ions, and it may be argued that the pH-values on the solid surfaces - which can be estimated from "pH-titration tests" - are decisive for the behavior of the particular element rather than the pH-values determined in solution.

Release Prognosis

The combination of mobilisation kinetics and the chemical metal forms enable to prognosticate the long-term release behaviour of metals from the landfill under estimation. For this purpose the "quick-motion" factor of the experimental design has to be evaluated.

For zinc under the flow-through conditions at pH 5 and 400 mV, from the total release (437 mg per kg) and the amount of water passing through the landfill material (140 l), an experimental zinc release of approx. 3 mg/l can be calculated (see table 2). The measured concentrations in the pore-water of the deposition site are between 0.04 mg Zn/l and 1.40 mg Zn/l, in the groundwater just beside the landfill approx. 0.02 mg Zn/l. If all this zinc in the nearby groundwater would have been resulted from the landfill (worst case assumption), a "black-box-experimental-factor" of 150 can be calculated from the ratio of the proposed and the real zinc concentrations.

If this factor is multiplied with the duration of the experiment (10 weeks), a time period of nearly 30 years is represented by the experiment. It can be concluded from figure 5 that the release of the rest of "oxalate-bound" zinc would take another 12 experimental weeks; this would be equivalent to nearly 40 years in nature. Altogether, a time period of at least 50 to 100 years would be covered by the experiment, and a prognosis within this time scale can be drawn from the present results.

The "quick-motion-factor" of 150 represents a minimum, because it is based on the assumption that all zinc measured in the groundwater in the immediate neighbourhood of the landfill is release from that site. However, similar zinc concentrations are very common in groundwater aquifers not affected significantly by anthropogenic activities. If significant amounts of zinc are not originating from the deposit, the experimental factor is higher and the time scale of prognosis, therefore, would be even longer. In any case, after this time period the zinc concentration in groundwater will decrease.

The upper limits of the prognosis can be evaluated by the proton balance because the pH-value is the most important parameter for the metal release in many such examples. The porewater of the waste material exhibits a pH-value of 6, i.e. the proton concentration is 10^{-3} mmol/l. During the experiment, conducted during a time period of 10 weeks, 25 mmol H^+/l have been added to maintain pH 5, equivalent to a time factor of 25.000; this results in an extrapolated time period of 5.000 years, represented by the experiments on *the basis of the proton balance.*

For the present example, however, even such mobilisation cannot be expected in reality, since there is no internal process - such as sulfide oxidation or acetate production from organic degradation - which could affect pH conditions in the deposits. Additional calculations on the basis of atmospheric inputs of acidity can be interpreted in a way that with the present form of deposition of the metal oxide residues from heat processing below the groundwater level and excluding degradable organic substances no significant change of metal mobility or release will take place with several thousand years.

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HUMIC SUBSTANCES FORMED BY MICROORGANISMS FROM MUNICIPAL REFUSE, LANDFILL LEAKAGE WATER, AND GROUNDWATER: SIMILARITIES AND DIFFERENCES

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Summary

The origin of humic substances in groundwater aquifers has not yet been completely elucidated. We have attempted to determine whether humic substances may be formed from simple non-humified organic materials by natural microbial populations associated with municipal refuse, landfill leakage water, and groundwater. In liquid cultures containing starch, casein, finely milled refuse or fossil plant material as substrates, different amounts of humic substances were formed by the natural communities of microorganisms. Carbon and nitrogen contents, spectroscopic characteristics in the UV and visible ranges of light as well as FTIR spectroscopy indicated similarties but also distinct differences in the structural composition between newly formed humic substances and those originating from groundwater.

INTRODUCTION

Humic substances contribute a share of up to 90% of the dissolved organic carbon in groundwater. Groundwater aquifers located near controlled tips usually contain high concentrations of dissolved organic carbon, the main portion of which belongs to humic substances (BARBER et al., 1981). The origin of these substances has remained doubtful. Soil overlaying groundwater aquifers may undoubtedly play a role as a source of water-soluble humic materials in groundwater. However, other sources, e.g. leachates from fossil carbon deposits such as kerogen should also be considered (THURMAN 1985). In our laboratory experiments, we attempted to estimate whether microorganisms from municipal refuse, landfill leakage water, and groundwater are capable of forming humic substances from simple organic precursors. The capability of groundwater microorganisms to form humic substances from fossil plant materials in groundwater was also investigated.

Materials and Methods

Samples of municipal refuse and landfill leakage water were collected from three large scale experimental tipping sites at Wolfsburg and Braunschweig (F.R.G.) as described by FILIP and KÜSTER (1979) and by SPILLMANN (1986). The material different in age and composition was designated as "rotted" or "compacted" (both aged for about 6 months), and "old" (decomposing more than 5 years). The groundwater samples were collected from a raw water deep-well (depth: 150 m) at Langen (F.R.G.). For microbiological characterization of the individual samples, cultivation techniques such as the Most Probable Numbers-method (MPN), and plate count were used. They were supported by electron microscopical investigations to obtain more information about microbial communities in the environmental samples (SMED-HILDMANN and FILIP 1988). The basic nutrient solution consisted of 0.5 g KNO_3, 1 g K_2HPO_4, 0.2 g $MgSO_4$, 0.1 g $CaCl_2$, 0.1 g $NaCl_2$, traces of $FeCl$, 0.5 g asparagine, and 1 g mannitol per litre. Alternatively, mannitol and KNO_3 were replaced by 30 g casein or 70 g or 35 g dry powdered refuse per litre, respectively. When adding starch (26 g/L), no mannitol but NH_4NO_3 (3 g/L) was used. Five mL of the 1:100 refuse samples diluted in sterile deionized water was used as inoculum for 300 mL culture solution; otherwise, an undiluted landfill leakage water (8.5 mL) was used per 500 mL nutrient solution. The groundwater cultures were prepared with 450 mL of the natural groundwater sample to which 50 mL adequately concentrated nutrient solution as above was added. To some groundwater samples, 5 g of sterilized powdered fossil plant material originating from a quaternary deposit (Aller Urstromtal) overlaying a groundwater aquifer at Fuhrberg near Hannover (F.R.G.) was added per 500 mL as sole nutrient source. At least three parallel cultures and control flasks were incubated for 60 days (inoculum:

refuse), and 90 days (inoculum: landfill leakage water), respectively at 35 oC in the dark as standing and shaked cultures. The cultures consisting of natural communities of groundwater microorganisms were kept standing for 6 months either at 35 oC or at 4 oC in the dark. When cultivation had terminated, humic acid from the liquid phase (HA1) or from the microbial biomass (HA2) were separated as described by FILIP et al. (1972).

The carbon contents in liquid and solid samples were estimated using the ASTRO TOC-TC Analyzer 1850 (U.S.A.), and by wet incineration after ALTEN et al. (1935), respectively. The total nitrogen was analyzed by the Kjeldahl method using a KJELTEC AUTO 1030 Analyzer (Tecator, Sweden).

The estimation of the total acidity of humic substances was carried out as described by SCHNITZER (1982).

The optical density of humic acids was measured over the wavelength ranges of 200 - 400 nm, and 400 - 800 nm in the recording spectrophotometer SPECTRO-A (AGW-Analysengeräte, F.R.G.). E4/E6 ratios were calculated from the extinction values at 465nm and 665nm (CHEN et al., 1977).

For Fourier transform infrared spectroscopy (FTIR) KI pellets containing 2% sample were pressed at 250 atm and examined in the BRUKER IFS 85 FTIR spectrophotometer over the wavelength range of 2.5 - 15 µm (4,000 - 300 cm E-1).

The reference humic acid originated from a groundwater aquifer located at Fuhrberg near Hannover (F.R.G.) and was obtained from the local water works.

Results and Discussion

Both municipal refuse from a landfill and the leakage water were rich in microorganisms (Table 1). Up to 10 E10 colony forming units (CFU) per 1 g dry weight or per 1 mL were counted. The microflora of the individual refuse samples differed, showing equal parts of gram-negative and gram-positive bacteria in the material from the rotted landfill, and almost only gram-negative bacteria in the old landfill. Microbial counts in the leakage water were similar to those obtained for refuse. In the groundwater samples, the total numbers of microorganisms were less than 10 CFU per 1 mL when estimated by cultivation techniques. Higher counts (up to 10 E5 per 1 mL) have been found by transmission electron microscopy.

TABLE 1

Numbers of Some Microorganisms in Refuse ($n \times g^{-1}$ dry matter) and in Leakage Water Samples ($n \times mL^{-1}$)

Microorganisms	Rotted		Compacted		Old	
	Refuse	Leakage Water	Refuse	Leakage Water	Refuse	Leakage Water
Aerobic Proteolytic Bacteria	1.8×10^{8}	1.9×10^{10}	1.2×10^{8}	1.4×10^{10}	4.2×10^{6}	4.3×10^{5}
Anaerobic Proteolytic Bacteria	2.0×10^{9}	1.1×10^{10}	2.5×10^{10}	1.1×10^{10}	4.4×10^{6}	4.5×10^{4}
Actinomycetes	7.3×10^{6}	4.0×10^{4}	2.6×10^{6}	$< 10^{4}$	1.9×10^{6}	$< 10^{4}$
Fungi	5.8×10^{6}	3.4×10^{6}	1.5×10^{6}	1.1×10^{9}	1.2×10^{5}	2.3×10^{9}

Cultures enriched with casein yielded the highest amounts of humic substances (Table 2). Although a rather strong microbial growth was observed in standing cultures enriched with starch and inoculated with leakage water, no humic substances (HA1) were isolated from the cultural liquids. Shake cultures of the same composition, however, mostly contained some humic substances. This may be due to a better oxygen supply in shake cultures. Aerobic conditions undoubtedly support mineralization of organic substrates, but they may also produce higher amounts of energy which can be used in the humification process (NOVAK 1963).

The groundwater cultures showed only a poor microbial growth. Some low yields of biomass were obtained but no humic substances. The microbial growth was better at 4 oC than at 35 oC. During the 6 month incubation period, no growth was observed in the cultures of groundwater microorganisms containing the fossil

TABLE 2 Formation of Humic Acids in Liquid Cultures Enriched with Casein, and Inoculated by Indigenous Microorganisms from Landfill Leakage Water Humic Acids in Liquid Phase (HA1) (mg/L), and in Microbial Biomass (HA2) (mg/g Dry Matter); Standing Cultures on Upper, Agitated on Lower Line.

Inoculum	ROTTED		COMPACTED		OLD	
Medium	HA1	HA2	HA1	HA2	HA1	HA2
Alone	0 0	0 0	0 0	0 0	0 0	0 0
Casein Add	103 606	23 14	263 486	4 29	83 848	26 36
Starch Add	0 0	5 12	0 212	14 29	0 100	7 7
Refuse Add	96 66	9 8	149 258	5 4	258 262	4 4

plant material. The groundwater microflora seems to consist of species with specific nutrient composition requirements. Its ability to form humic substances also seems to be very limited. Further attemps have to be made to elucidate the characteristics of microorganisms in groundwaters in more detail.

The C and N contents of the humic acids formed by microbial activity are similar to those refered by other authors (MATSCHKE 1970, GONZALEZ-VILA and MARTIN 1985), i.e., lying below 50% for carbon and below 5% for nitrogen.

The total acidity (exchange capacity) amounted to 9.33 meq/g for groundwater humic acid, and 11.80 meq/g for the microbially formed humic acids. For the humic acids taken from the municipal refuse and from the landfill leakage water, values of 5.61 meq/g and 5.85 meq/g, respectively, were estimated.

The visible spectra of some humic preparations are given in Fig.1. The slopes of the absorption curves indicate a relatively low optical density for the materials which originated from the microbial cultures in comparison to those from municipal refuse, groundwater, and leakage water. The UV spectra given in Fig.2 show more similarities. A major feature appears at 275 - 280 nm indicating the presence of soluble phenolics and quinolics (WILLIAMS and FLEMING 1971). FILIP et al. (1976) found similar absorption maxima in melanins formed by different soil fungi. The respective spectra of the humic acids from municipal refuse and groundwater were featureless. The microbial humic substances seem to represent "young" humic acids which may undergo further ageing and polymerization. As reported by NEWMAN et al. (1987), the aromaticity of the refuse-related humic acids increases with time of refuse disposal.

The values of E4/E6 ratio were about 6 for most of the humic acids tested. According to CHEN et al. (1977), values in this range, except for soil humic acids are associated with a relatively high molecular weight or large particle size of humic substances.

The FTIR spectra of the humic substances are shown in Fig.3. They indicate an almost identical structural composition of humic materials from the individual microbial cultures. On the other hand, distinct differences exist between these materials and groundwater humic acid. The notable features in the groundwater humic acid are the major O-H streching bands in the 3,400 cm E-1 region, C = O bonding of carboxyl and carbonyl groups at 1,718 cm E-1, 1,624 cm E-1, and 1,225 cm E-1. At the 1,225 cm E-1, phenolic OH groups also absorb. C - H bonds of methyl and/or methylene groups absorb rather weakly at 2,972 cm E-1 and 1,388 cm E-1. A shoulder at 2,600 cm E-1 can be attributed to O - H streching in carboxyl groups. The humic preparations from the individual microbial cultures show more differentiated FTIR spectra. The major absorption at 3,300 cm E-1 with shoulders at 3,400 cm E-1 and 3,100 cm E-1 indicates not only the O - H streching bonds but also the presence of NH groups. Amide groups were clearly demonstrated at 1,653 cm E-1, 1,539 cm E-1, and 1,233 cm E-1 (amides I, II, III). Thus, these preparations consist of a higher proportion of peptidic structures or amino acids which may originate from microbial biomass or residual substrate (casein). Clearly distinguishable absorption in the regions of 2,960 cm E-1 - 2,855 cm E-1, and also at 1,454 cm E-1 indicates the presence of CH2 and CH3 groups in aliphatic compounds. A C - O strech in some aromatics and even more in alcohols and/or in polysaccharides appears at 1,046 cm E-1. Alcoholic O - H groups are indicated also by a shoulder at 1,159 cm E-1. Thus, the FTIR, UV and visible spectra all indicate a higher content of

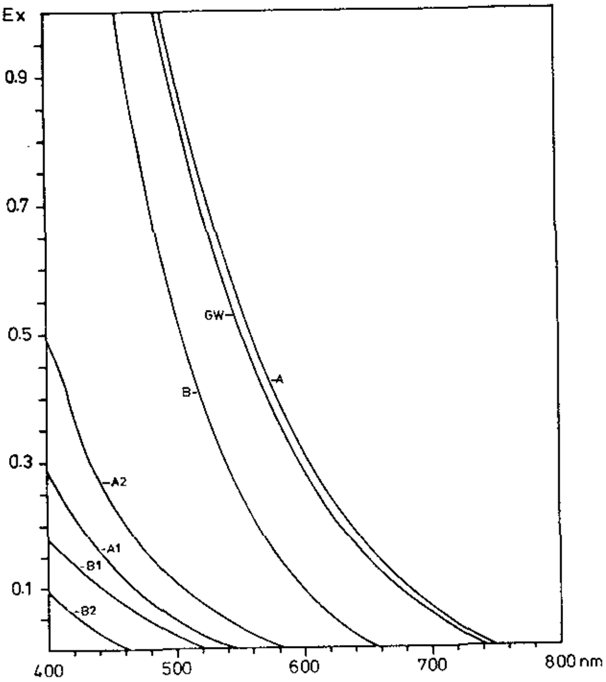
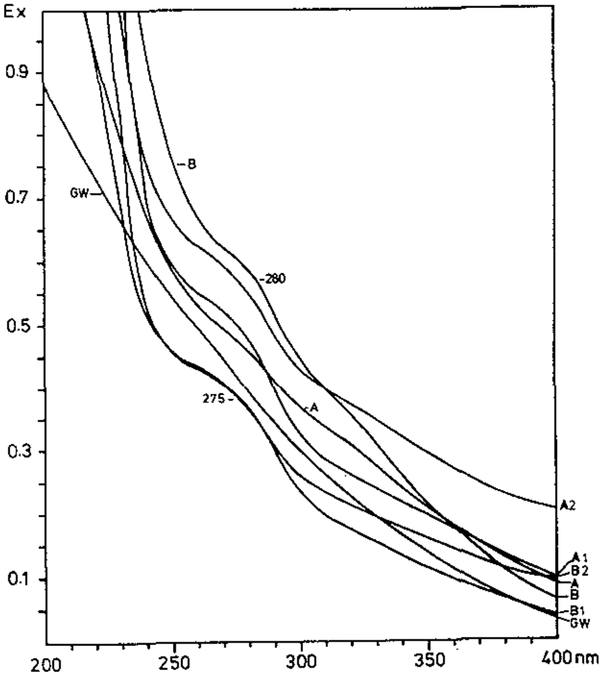

Ex
0.9
0.7
0.5
0.3
0.1
GW
A
B
A2
A1
B1
B2
400
500
600
700
800 nm

Fig.1


Ex
0.9
0.7
0.5
0.3
0.1
B
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280
275
A
A2
A1
B2
A
B
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GW
200
250
300
350
400 nm

Fig.2

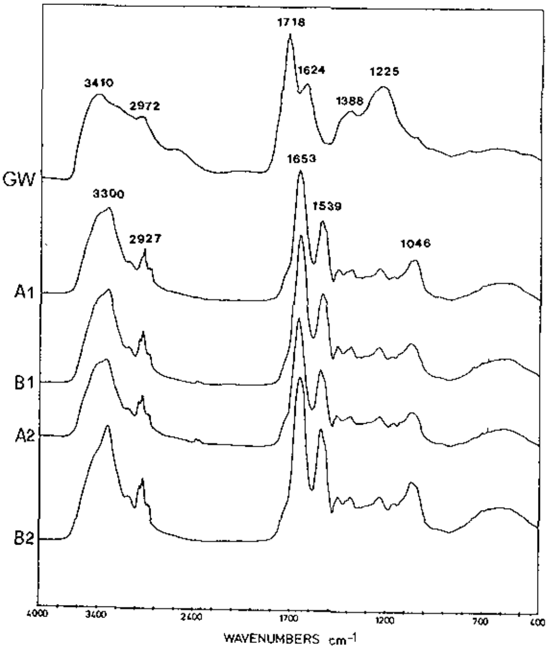

Fig.3

Figures

Visible (Fig.1), UV (Fig.2), and FTIR spectra (Fig.3) of selected humic acids (HA). A = HA from rotted refuse; A1 = HA from the liquid phase of standing cultures enriched in casein, and inoculated with an infusion of rotted refuse; A2 = same as A1 but shaked cultures; B = HA from rotted landfill leakage water; B1 = HA from the liquid phase of the standing cultures enriched in casein, and inoculated with an infusion of rotted landfill leakage water; B2 = same as B1 but shaked cultures; GW = HA from groundwater.

aliphatic compounds, and especially NH-units in the microbial humic substances. Similar features were observed in our previous studies dealing with humic-like polymers newly formed from simple organic substrates by microorganisms (FILIP et al., 1976).

CONCLUSIONS

The results obtained in this study indicate the capability of the microflora indigenous to municipal refuse to form humic substances from simple organic substrates. These humic substances are of aliphatic nature, containing a high proportion of nitrogen. Transport of humic substances into groundwater aquifer can be assumed rather than their direkt formation in groundwater. Humic acid from groundwater appears to be higher polymerized, and contains more aromatic structures, due to a long-term ageing process in the underground.

Acknowledgement

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SHALLOW-LAND DISPOSAL OF LOW-LEVEL RADIOACTIVE WASTE

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Summary

The rationale for disposal of low-level radioactive waste is to isolate it from the biosphere until radioactive decay renders the radionuclides no longer hazardous to life. Low-level radioactive waste is commonly disposed at shallow depths with various engineered features to stabilize the waste and to reduce its dissolution and transport by water. The unsaturated zone is generally preferred for disposing of low-level waste. Low-level radioactive waste may need to be isolated for 300 to 500 years. Engineered barriers are not relied upon to isolate the waste for their hazardous life. Therefore, a combination of engineered barriers and the geohydrology of the disposal site needs to provide isolation of the waste for the hazardous life of the radioactive waste.

INTRODUCTION

Low-level radioactive wastes are composed primarily of nuclides of relatively short half-lives and of low radioactivity. Low-level radioactive wastes are produced in the nuclear-fuel production cycle as well as from the use of radionuclides in research, medicine, and industry. The isolation time required for a waste is a function of the half-lives of the radionuclides and their daughter products in the waste. A rule of thumb commonly used in the nuclear industry is that radioactive waste should be confined at least 10 times the half-life of the longest dominant isotopes. The half-lives of principal components of low-level radioactive waste range from 3 to 30 years. For waste containing primarily strontium-90 and cesium-137, 300 years of isolation (about 10 half-lives) is required to reduce the activity to one-ten thousandth of its initial activity. Thus, on the basis of half-lives and concentration limits, the waste should be isolated from the biosphere for 300 to 500 years.

Many disposal methods have been proposed or used in efforts to isolate low-level radioactive waste from the biosphere during its hazardous life. Disposal facilities are generally near-surface, within 50 meters of the land surface with or without added engineering barriers. They are of several types, but the burial of wastes is commonly in trenches. Alternative methods include burial in pits and shafts, and the placement of waste in engineered near-surface facilities such as earth-mounded concrete bunkers, lined shafts, below- and above-ground vaults, and mined and natural cavities. A summary of shallow-land trench burial sites in the United States is given in Fischer and Robertson (1984), Bedinger and Stevens (in press), and Bedinger (in press). A summary of alternative methods for disposal of low-level radioactive waste used in various parts of the world is given in Bedinger (1987). The following discussion will be concerned primarily with the geohydrologic aspects of shallow-land disposal of low-level radioactive waste.

SHALLOW-LAND TRENCH BURIAL OF LOW-LEVEL RADIOACTIVE WASTE

The most likely path of nuclides from a shallow-land radioactive-waste repository to the biosphere is transport by water, but shallow repositories are also subject to intrusion by man, plant roots, and burrowing animals; exposure by erosion; and migration of radionuclides as gas in the unsaturated zone. Successful disposal of low-level waste depends upon application of many scientific and engineering disciplines to provide site selection and design and also the effective engineering construction and operations at the site. Examination of the performance of low-level radioactive-waste sites, some aspects of which are summarized below, shows that whereas many sites exhibit satisfactory aspects in the isolation of radionuclides, the coordination between scientific disciplines and engineering has not been accomplished to the point where acceptable siting, design, and operation of a low-level radioactive-waste site is a predictable or routine procedure.

Many of the first generation of low-level radioactive waste disposal sites in the Unite States were excavated in deposits of low permeability. Such siting resulted in accumulation of water in the trenches, and saturation and accelerated leaching of the waste. This phenomenon, referred to as the "bathtub effect," has occurred at low-level radioactive disposal sites at West Valley, New York; Maxey Flats, Kentucky; and Oak Ridge National Laboratory, Tennessee. Disposal of organic hazardous wastes may be disposed in trenches with impermeable bottom membranes and the accumulated water pumped out and treated or recycled and the waste environment aerated or treated chemically to promote disintegration of the waste. In contrast, radioactive disintegration cannot be accelerated. Excessive accumulation of leachate in radioactive waste trenches would require long time periods of active pumping and volume reduction treatment to assure isolation of the waste until it is no longer hazardous.

Other problems which developed at low-level radioactive waste sites included collapse of trench caps and migration of radionuclides from the burial site. Collapse of trench caps has been caused principally by compaction of the waste packages and backfill material. Documentation of trench cap collapse has been most completely documented by Gray (1988) at the burial at site Sheffield, Illinois. Efforts to mitigate infiltration through the trench caps and collapse are reported at West Valley New York, by Prudic (1986) and at Barnwell, South Carolina, by Dennehy and McMahon (1985). At West Valley, New York, limited success was achieved by compacting the clay capping materials with heavy machinery and applying an overburden of earth material for a limited period of time. At Barnwell, South Carolina, the waste packages are stacked systematically to allow backfill to fill the voids; the backfill material is sand which minimizes compaction of the backfill material.

HYDROLOGY OF SHALLOW-LAND DISPOSAL

The experience in the United States with shallow-land trench disposal operations has provided some lessons that could be applied to the next generation of low-level radioactive waste repositories. Problems of waste containment in the unsaturated zone are largely problems of maintaining a minimum of contact between the waste and water. Climate and climate-related factors are undoubtedly the most important hydrologic factors affecting the suitability of a site for shallow-land disposal. Climate influences in large measure the infiltration of water into a repository, the thickness of the unsaturated zone, the stream density and hence the distance from a repository to the ground-water discharge point. For example, compare the annual precipitation, depth to water, recharge, and ground water travel time from the site to discharge area at an arid site, Beatty, Nevada, from Nichols (1985) and Bedinger *et al.* (1984) and at a humid site, Barnwell, South Carolina, from Cahill (1982) and Dennehy and McMahon (1985).

	Arid Site	Humid Site
Annual Precipitation (mm/yr)	114	1,170
Depth to water (m)	85-115	1,067
Recharge (mm/yr)	.04	380
Ground water travel time (yr)	500-1000	50

It is conceded that the waste cannot be retained perpetually within the repository. Hydraulic barriers, both natural and engineered, are employed in design of trench repositories to minimize the transport of radionuclides from the repository. In the unsaturated zone, flow may be retarded by capillary barriers in which a fine-grained layer overlies a coarse-grained layer. The barrier is created by the lower permeability of the coarse-grained layer at high moisture tension underlying a more permeable fine-grained bed.

A clay cap over the trench repository provides both one component of capillary barrier and a low permeability barrier to infiltration of precipitation. Reed (in press) points out that in humid regions where capillary barriers may not be effective, the most important function of the clay cap is to limit infiltration.

Low permeability materials are severely limited as suitable host media for low-level radioactive waste repositories in the unsaturated zone. Capping materials are not impermeable and experience shows that they will become more permeable with time due to weathering, desiccation and biologic activity. The resulting phenomenon is increased infiltration and accumulation of water in the trench because the low permeability host media will not permit the repository to drain as rapidly as water infiltrates the repository. Meyer (1979) questioned if a more permeable host medium would not avoid the problems of the bathtub effect. Fischer and Robertson (1984) recommended that the host medium be permeable. Though the severe restrictions of low permeability material as a suitable host media have been amply demonstrated by the first generation of trench repositories, recent screening studies for low-level radioactive waste sites have been conducted that have targeted low permeability material as the host medium.

Barriers to radionuclide migration should also be present in the ground-water flow path in the region between the repository and the natural discharge area. Factors which present major barriers to radionuclide transport include: (1) Long flow paths, low hydraulic gradients and substantial effective porosity, and (2) those which decrease concentration of radionuclides in solution, such as sorption and minimal solubility of waste. Dispersion, diffusion, and dilution are processes which also reduce point concentrations of contaminants but do not reduce the amount of contaminant in solution.

There are neither experimental nor real time bases for long-term projection of the effectiveness of engineered barriers. Engineered barriers may serve a useful purpose for a limited time, but they should not be designed in any way that would ultimately reduce the effectiveness of the natural hydrologic system in isolating waste from the environment. Engineering barriers, including those designed to isolate the waste, drain the repository, stabilize the waste or the repository, or prevent the waste from coming into contact with moisture cannot be relied upon to provide isolation for the hazardous life of the radioactive waste.

Selection of sites and design and operation of suitable low-level radioactive waste repositories requires a combination of scientific and engineering application. Many lessons have been learned from the study of existing burial sites and the application of geologic and hydrologic principles. It remains to be seen by the performance of the next generation of repositories if the lessons have been properly interpreted.

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Environmental aspects and cases

MIGRATION OF INDICATOR MICROBES FROM LEACHATE IN AN ARTIFICIAL AQUIFER

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SUMMARY

The survival time and the kinetic of E. coli as the indicator organism for fecal pollution were tested in an artificial aquifer contaminated by leachate. It was shown that the pathogenic bacteria survived approximately 130 d and moved around 30 m away from the inoculation point, when the aquifer was contaminated only once. By a continous contamination over 5 months we could isolate E. coli for 500 d. The bacteria moved 55 m away.

INTRODUCTION

One possibility to get rid of municipal waste is to dump it in a landfill. Concerning the law and the experiences during the last years, we believe that dumping in a so-called controlled landfill is not dangerous to the adjacent soil and groundwater. Actually every dumping of waste can influence the air, soil and groundwater.

Untill this year we are sure that a basal cointment of clay by a k_f-value 10^{-9} m/sec is safe enough that no leachate or bacteria can percolate into the groundwater. Recent works however showed that especially sodium clays are destroyed by leachates. Therefore it is possible that toxic substances and bacteria penetrate into the groundwater below a landfill, poisoning the resource which we use as drinking water.

Because of the practice to deposit houshold refuse together with sewage sludge it is possible, that pathogenic bacteria penetrate into the groundwater and perhaps into the wells which are used for the drinking water supply.

Besides it is necessary to evaluate a threat of fecal pollutions which are spread on an unknown covered deposit or a recultivated one. The mentioned contamination can occur, for example when an area is used as a resource center, animal hence agricultural land or by wild living animals.

The leachate of decades of old dumpings can still contaminate the underground soil and can percolate together with pathogenic bacteria from the surface into an aquifer (DFG, 1986).

The occurence of pathogenic bacteria in the leachate does not depend on the handling of the dumping. We could isolate our test bacteria E. coli in the outflow from anaerobic pressed waste sides or aerobic ones.

It is known that the transportation of the bacteria depends on the flowing speed of the ground water. In a clay ground it runs some few centimeters, in a sand about 1 meter and in debris many kilometers per day. The survival time of E. coli in a pure water for example runs up to about 300 days (Havemeister,1983).

Because of those data, we can approximately calculate how long the pathogenic bacteria survive and how far they move. We can also estimate when one can use the water from a contaminated aquifer as a resource for drinking water.

The effects of the leachate of dumpings on the ground water are very specific. One can not predict the volume or the chemical and the microbial composition of the leachate (DFG, 1986).

With a project promoted by the DFG we wanted to check the following questions.

Persistence of the test bacteria at the inoculation point and in the aquifer by a solitary and a continous contamination

Kinetic of the test bacteria by a solitary and a continous contamination

Rate of spreading of the bacteria with regard to the flowing speed of the water

METHODS

We have built up an artificial aquifer which is flowed through by a mixture of groundwater and leachate. The aquifer filled with sand is 100 m long and the diameter 1 m. At 14 different points distributed over the 100 m long aquifer one can take the water sample through pipes by a vacuum pump (Figure 1, DFG, 1988). We examined 100 ml samples, correspondingly the limiting value of the German Drinking Water Regulation.

The tested leachates were gathered from an operating landfill. To simulate in situ conditions we accepted that the chemical composition of the leachates differs over the test periode. This means that the results which we get are not from exactly parallel tests.

For the tests we used E. coli, which is the indicator for fecal pollution respectively for Salmonella. Streptococci, Proteus and Pseudomonades as indicators for environmental pollutions were also tested.

The aquifer was inoculated by different amounts of bacteria from 10^9 - 10^{12} cfu (colony forming units). We infiltrated the aquifer once or several times. In the latter case 3 times a week for 5 months.

RESULTS

Aquifer without leachate

We found that the bacteria moved between 10 and 35 m. In the tests with the continous inoculation the migration distance was not significantly longer. The spreading velocity amounts around 0,20 - 1 m per day. The test bacteria survived at the inoculation point between 62 - 132 d.

Aquifer contaminated with leachate

The migration distance amounts between 5 m and 45 m by the solitary and 26 - 55 m by the continous inoculation. The survival time at the inoculation point was around 73 - 105 d aerob leachate and 114 - 134 d anaerob leachate by the solitary, 250 - 500 d by the continous inoculation respectively. The spreading velocity was comperable to the flowing speed of the water. There was no difference in the survival time and migration distance by an inoculation with 10^8 - 10^{12} cfu of the bacteria. E. coli and Proteus gave more or less the same results. The Pseudomonades survived a little longer and moved farther. We also found that a combination of those bacteria in a test led to an earlier elimination of E. coli and Proteus. Whereas the Pseudomonades can be isolated longer. The Streptococci survived 200 d and moved around 70 m.

We als found that the bacterias did not contaminate all levels of the aquifer (Figure 2).

CONCLUSIONS

The both tests with the groundwater and the mixture of groundwater and leachate show nearly the same results. The migration distance was between a few meters and 30 m, the persistence about 70 - 130 d. Those data point out that the added leachate hardly influences the behavior of the bacteria in a dilution 1 : 10 (leachate/groundwater).

In the test with continous contamination it was shown that the inoculated amound of test organisms did not effect the survival time and the kinetic so much. More important is the length of the period where the bacteria penetrate into the aquifer. This means that small amounts of pathogenic bacteria inoculated over a long time have a worse effect than large amounts added only once. Very interesting was the fact that the added bacterias do not move togeteher in the shape of a plug through the aquifer becoming more and more diluted (Figure 3, Knoll, 1982)

Actually most of the bacterias stay at the inoculation point and only some are flowed away till to a maximum distance. After their reaching this point which is between a few meters and 50 m away, the bacterias move back to the inoculation point. Some time later one can only isolate the pathogenic bacteria at the inoculation point, where they persist over a long time. We also found in some cases that the contaminated aquifer was suddenly free from the test bacteria.

At the moment we can not explain this behavior but we think that changes of the pH-value and different chemicals can inactivate the bacteria or may also cause them to start to move again through the aquifer.

We also have to mention that the resistance of the different bacteria especially Salmonella has changed, because of the polluted environment. This means that the long known standards are possibly not safe enough any more (Mersch-Sundermann, 1987).

The present project has shown that the leachates from dumpings do not promote the survival of the test bacteria. They do not survive longer compaired with the data from pure ground water tests (Althaus,1983). Significant differences between aerobic and anaerobic leachates could not be proved. It seems however that E. coli survived a little longer in a mixture with anaerobic leachates.

Because of the great deviations between the results of the parallel tests, we think that one can not build a formula for calculating the elimination of the pathogenic bacteria in a contaminated aquifer. Other authers gave an elimination constant for pure groundwater (Pekdeger, 1983). Possibly one can imagine that those standards will cause that the accidents are calculated and evaluated in a wrong way.

With our work we could show that the different pollution factors and the leachates influence the anthropogenic bacteria in various ways. We also have to consider the soil matrix - sand, clay, humus and other organic substances. Pathogenic bacteria are absorbed by those substances in different ways. Therefore we have to consider that we can possibly get varying survival times and migration distances. This means that our test gave only data about sandy aquifers which do not consist of larger parts of clay and humus.

When one assesses the results we can recommend that the old so-called 50-day-line from Knorr (Knorr, 1951) is also a standard for the evaluation of an accident by leachate together with pathogenic bacteria. We could show that the fecal bacteria were more or less eliminated in the area which the water reached after 50 days. But we have to keep in mind that the contamination point can strew pathogenic bacteria for several hundred days after the actual contamination. Therefore we have to calculate a risk of infection for a long time.

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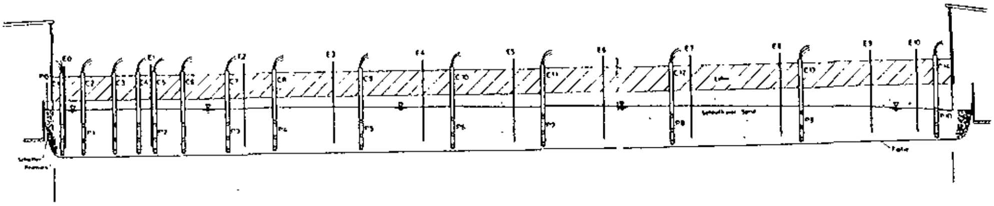
Figure 1

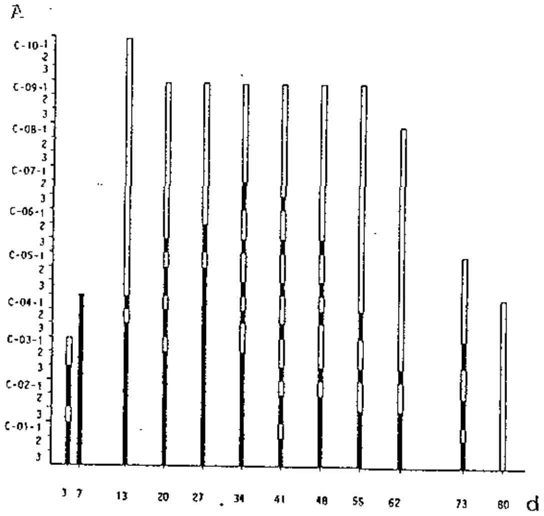
Artificial Aquifer



C-01 - C-10 Different sampling points and levels

Figure 2

Scattering into the aquifer

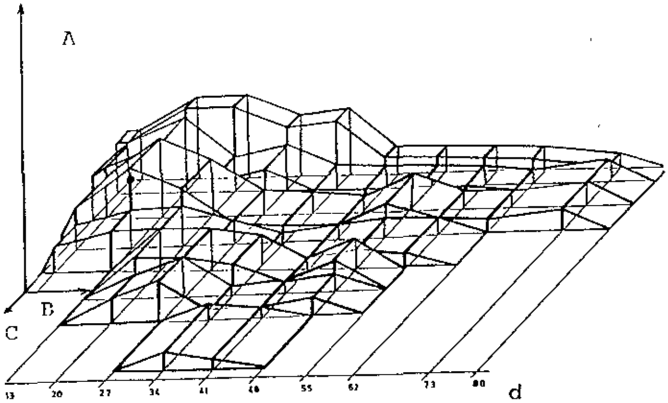


A - Sampling point

1, 2, 3 - different levels

Figure 3

Spreading of the bacteria



A - concentration

B - Time

c - Distance

synoptical time-distance-concentration diagramm

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STANDARDIZED EVALUATION OF GROUNDWATER CONTAMINATION BY WASTE DISPOSAL SITES

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Summary

A concept for the evaluation of groundwater contamination by waste disposal sites on a broad backgound of information is suggested and presented for discussion. It offers an aid to assess such types of contaminations on a sound scientific basis. All essential parts for this kind of evaluation, namely "concentration", "groundwater currency" and "toxicity", are standardized and evaluated for themselves and then associated to a new "integrated evaluation class". In connection with a likewise standardized "exposure assessment" and by completion of data it seems to be practicable in risk potential estimation of existing contaminant sources and for finding remedial action priorities.

INTRODUCTION

Groundwater contamination by abandoned waste disposal sites is of outstanding importance, because groundwater is the main resource of drinking water in the Federal Republic of Germany. Therefore, following the existing laws, the quality of groundwater must be saved, monitored and the existing unacceptable contaminations must be sanitized. This can be achieved by a thorough evaluation of a special contamination considering substance-specific and site-(exposure/usage)-specific parameters.

DETECTION OF GROUNDWATER-ENDANGERING WASTE SITES

The great number of (abandoned) waste disposal sites and other contaminated sites in the Federal Republic of Germany made it necessary to develop a hierarchically structured method which starts with few time- and costsparing investigations applicable to many sites with the purpose to select the hazardous ones. At these, further and more detailed investigations should follow to complete the characterization of the detected contamination (Kerndorff et al., 1985; Arneth et al., 1986; Brill et al., 1986).

The structure of the method is shown in Fig. 1. After the preinvestigation, screening and analyses of hazardous contaminants, the problematic sites can be identified. This is of the most importance, because only these have to be analyzed in greater detail (3. analytical step). The next step is the evaluation of the analytical results of the investigated groundwater contamination.

THE EVALUATION CONCEPT

The evaluation concept is suggested to assess negative *influences on man* and environment on the groundwater path by contaminants from waste sites being. It

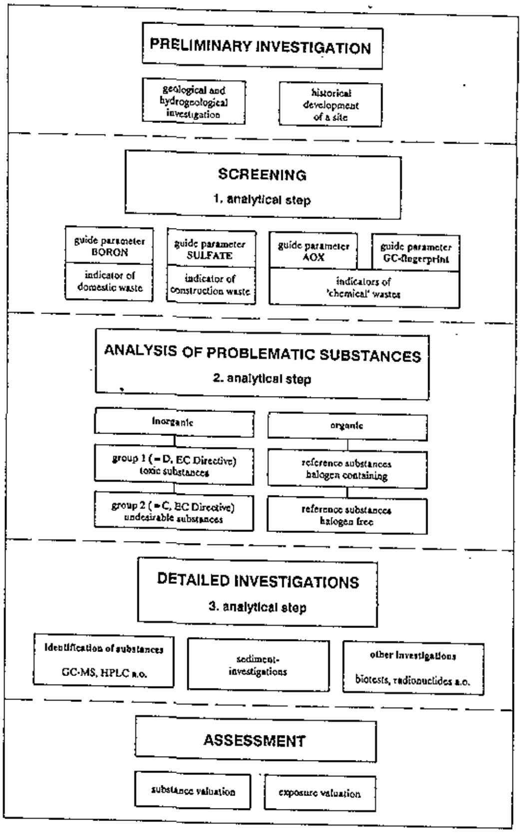


Figure 1:
Method for the detection of groundwater-endangering abandoned waste disposal sites

is devided into "substance valuation" and "exposure assessment" (Fig. 2). The "substance valuation" is subdivided into "concentration", "physico-chemical behaviour on the groundwater path" and "toxicity". Each part is evaluated individually and results are numbers between zero and 100. In the cases of "concentration" and "physico-chemical behaviour on the groundwater path", the parameter-specific representative spectrum of the measured values is brought into a cumulative curve. The value of the percentile is equivalent to the valuation number. In the case of toxicity, special kinds of procedures and parameter connections had to be elaborated.

The association of the three valuation numbers resulting from these procedures ("toxicity valuation number", "valuation number of groundwater currency" and "concentration valuation number"), flows - as an integrated valuation number - combined with the "exposure assessment" into an integrated evaluation scheme from which, on a sound scientific base, activities may derive.

The "Substance Valuation"

This part of the concept is divided into three areas considering path-specific aspects. These are "the physico-chemical behaviour of contaminants in groundwater", their "toxicity" and "the representative frequency distribution of their concentrations in a special area", for example in the Federal Republic of Germany, a province etc.

Elaboration of "Main Contaminants" and "Priority Pollutants"

In a laboratory routine it is not possible to analyze for all substances ever detected in contaminated groundwater. There must be a selection of substances which are to be determined and evaluated prior to others. This is an important aspect of the entire evaluation concept but cannot be dealt with in detail at this place. In the following paragraph, only a brief overview is presented.

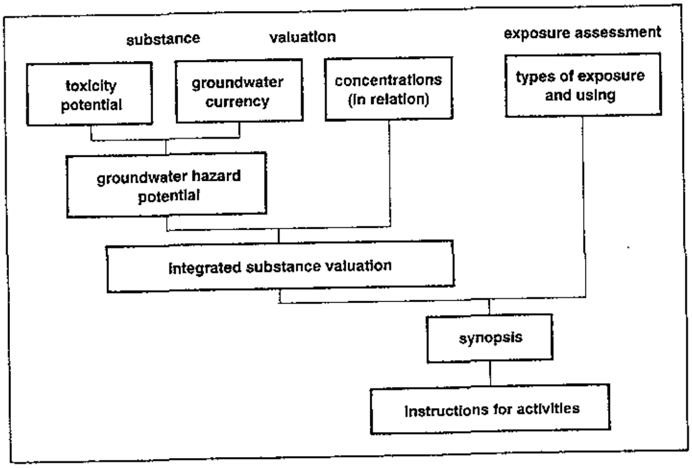


Figure 2:
Flowchart of the evaluation concept

Due to a different genesis of waste deposits and contaminated sites (plants etc.), different kinds of selection procedures for relevant parameters must be applied. In addition, the selection procedures should be path-specific and results based on a large statistic mass of measurements. With regard to the sequence, the most important criterion of selection, especially in the case of waste deposits, is the frequency of their detection in contaminated groundwaters (Plumb Jr., 1985, 1987; Plumb Jr. & Pitchford, 1985). For the following evaluation our own data from 60 sites and 40 sites from different provinces of the Federal Republic of Germany and Berlin (West) were available as well as a large data set from 500 sites in the USA. In total, about "only" 1000 groundwater contaminants could be detected downgradient of all these sites. Compared to estimations proceeding on the assumption that some 10^6 substances may occur in waste deposits is this a small number. With regard to a minimum detection frequency of 0.2% and a minimum concentration of 1 µg/l the number of substances is reduced to about 100. These are defined as main contaminants. In consideration of their physico-chemical behaviour, their toxicity and their concentrations substances with a high groundwater hazard potential will be selected. These are defined as "priority pollutants". In the future, analytical attempts should focus on these substances. Therefore, in our further research data of priority pollutants are to be collected and/or elaborated for establishing a list and to fill out the evaluation concept.

Representative spectra from concentrations of main contaminants

For the registration of background values and representative distributions of concentrations of main contaminants from groundwaters downgradient of waste sites in the area of the Federal Republic of Germany, large sets of data are necessary. Not all of these can be elaborated in one research project, so data from other sources must also be included. Such a data collection is presently done with the friendly help of many public authorities in the Federal Republic and Berlin (West). Analytical results concerning organic substances in groundwater contamination are not yet readily available. For this reason, an important aspect of the research program is to redress this deficit.

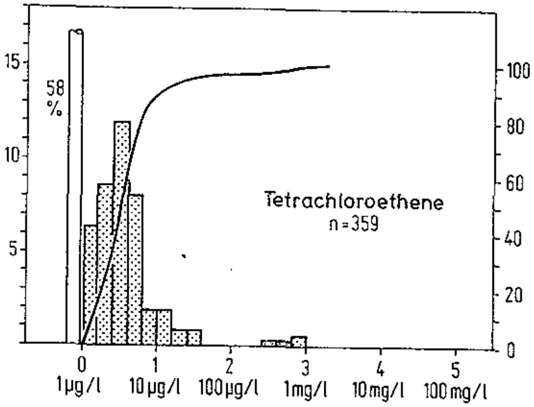


Figure 3:
Frequency distribution and cumulative curve of concentrations from Tetrachloroethene downstream of 92 waste sites in the Federal Republic of Germany

The frequency distribution of the concentrations of a substance is characteristic for a special area (FRG). This is demonstrated in Fig. 3 for tetrachloroethene. The results are positively skewed distributions due to comparably lower numbers of serious contaminations. For the specific evaluation of a measured concentration the cumulative curves are used. The percentile is defined as the valuation number. The cumulative curve is individual for each substance so that the evaluation is substance-specific and results in standardized valuation numbers between zero and 100.

Physico-chemical parameters - selection, quantification and connection

The occurrence of substances in groundwater (groundwater currency) is dependent mainly on factors as mobility and capability of accumulation. Each of these can be described by a different set of parameters or parameter connections. As an example, mobility can be described by water solubility, vapour pressure, boiling point, evaporation number, Ostwald's solubility, volatility from watery solution, adsorbability, density, viscosity, dissociation constant, surface tension, lipid solubility, Henry's law constant and others. The fact that many of these parameters are functionally dependent on each other, correlative or overlapping in their meaning (e.g. vapour pressure and boiling point) in addition to a non-sufficient availability of data for many parameters, makes it necessary in view of quantification to focus on some important ones of which nearly all data are available. This is the case for water solubility and vapour pressure.

Likewise, the capability of accumulation can be described, for example, the bioaccumulation by the partition coefficient of octanol-water (P_{ow}) the sorption affinity in soil (geo-accumulation) by the ^{1}chi-index (Sabljic, 1987). It is more difficult in the case of persistency because for most of the significant parameters no data are available to date. This is especially true for hydrolysis or half-life periods of aerobic or anaerobic metabolism. This is the reason while it is presently not useful to look further on this parameter.

The transport of solved contaminants with the groundwater stream is dependent on the capability of substances to be mobile ("mobility potential"). So, this parameter gives the tendency of a contaminant transfer on the groundwater path. Therefore, as an example, the evaluation of the mobility potential of a substance will be presented. Of the presently available main contaminants (ca. 100) derived from groundwater analyses downgradient of 92 waste sites, data for the parameters water solubility and vapour pressure were taken from literature. The resulting spectra are shown in frequency distributions and cumulative curves (Fig. 4). An evaluation number between zero and 100 derives from the cumulative curves for every substance which is equivalent to the percentile. For each parameter, that is water solubility and vapour pressure, the graph is individual. Each of the 100 contaminants has a fixed position on the cumulative curve which permits a parameter-specific evaluation. The connection of in this way standardized parameters vapour pressure (vp) and water solubility (ws) results in a quantified evaluation of the "mobility potential". This is shown in Fig. 4 for trichloroethene, tetrachloroethene and di-n-butylphthalate. The arithmetic mean of the two valuation numbers is calculated and defined as the "mobility valuation number". For "tri" it can be calculated as follows: (82(vp) + 62(ws))/2 = 72. For "tetra" it is 60.5 and for di-n-butylphthalate 22.

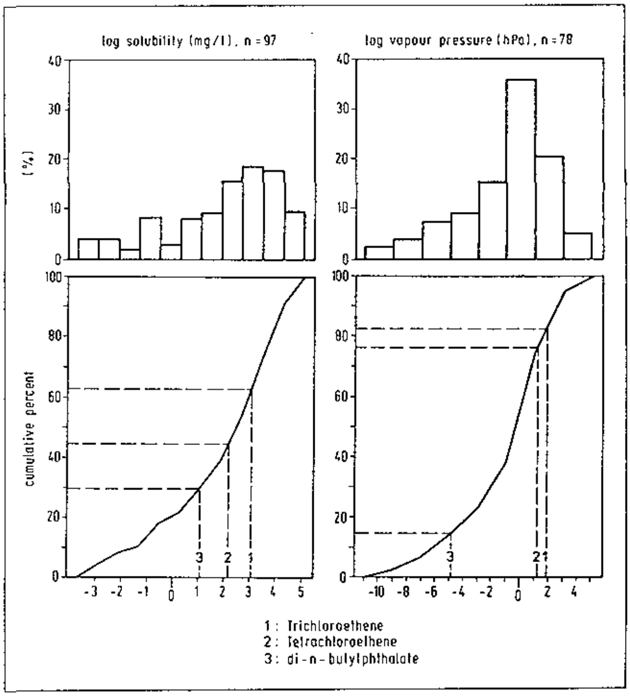


Figure 4:
Frequency distributions and cumulative curves of water solubility and vapour pressure from 100 main contaminants

Substances with mobility valuation numbers of more than 90 are assumed to have a high groundwater mobility potential, and those with less than 25 a low one. Due to an increasing amount of data, it can be expected that in the near future the gaps can be filled so that standardized evaluation numbers for the potential of a substance to occur on the groundwater path can be calculated more precisely.

The hazard potential of groundwater contaminants downstream of problematic waste sites due to their toxicity is looked at unsystematically, fragmentarily and only in special cases. The gauges used are for instance the ADI-Values (Acceptable Daily Intake) of the WHO or the comparable DTA-Values ("Duldbare tägliche Aufnahme") (DFG, 1986). They derive from the highest doses with no observable effect level (NOEL) under consideration of other toxicological parameters and by division of the NOEL with a "security factor". This is done by expert groups and is supposed to minimize insecurities coming from the nontransferabilities of animal experiments to man.

The ADI-Model actually cannot be used for a toxicological assessment of contaminants in groundwater because the ADI-values are established only for compounds such as pesticides and similar substances, that means, no data are available for groundwater contaminants from waste sites. For "detectable" or "questionable" carcinogenicity of substances no ADI-values are elaborated if an effect-free dose is not existent or if the mechanism of carcinogenicity is not exactly known. Because of significant gaps in toxicological evaluation of relevant groundwater contaminations an attempt is made to establish a toxicological evaluation model.

It takes care of these facts and specifically fits to the problem. The oral toxicity is the main aspect in this model because it considers the exposure-hazard over drinking water to be the most important. Inhalation or dermal intake must only be considered in special cases. The model reveals an "toxicty valuation number" of the "toxicological potential" of a substance between zero and 100 which may be combined with other relevant parameters. However, the model was not created to substitute the describing toxicological evaluation by corresponding administrative bodies.

In respect to the heterogeneity of toxicological data, the evaluation model is devided into five panels, "Rel" (Reliability of the tested criteria), "Tox" (Toxicological tests), "Ca" (Carcinogenicity), "Ott" (Other toxicological tests) and "Htx" (Human toxicology) (Fig. 5).

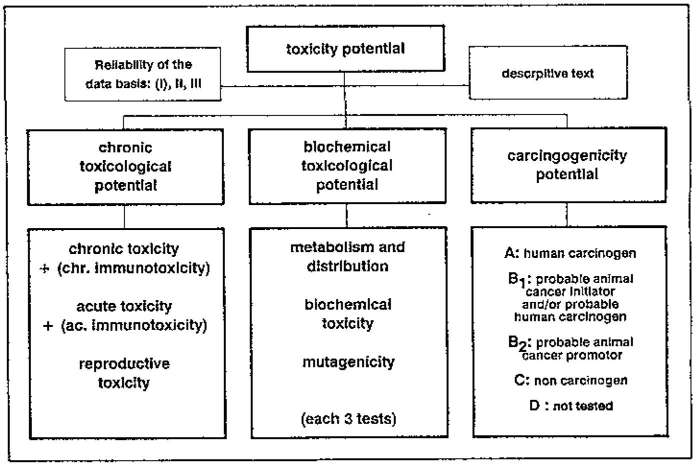
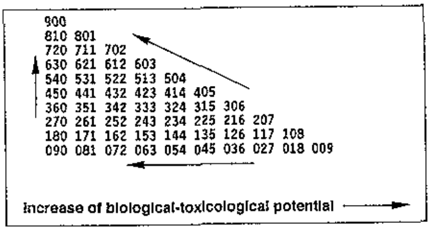


Figure 5:
The toxicological evaluation model for groundwater contaminants

According to the completeness the panel "Rel" (reliability) gives a general view over presently available data and thus determines the value of a toxicological substance assessment. The method should not be applied, however, to substances with a very incomplete set of data (group I). The panel "Htx" (human toxicology) was created for toxicological descriptions and cannot be considered in quantifications.

The "valuation numbers" deriving from the quantitative "Tox" panel and the qualitative "Ott" panel are used for further calculations. The weighed dose-number (tox-panel) remains unchanged, in contrast to the "Ott"-valuation number which must be simplified. The sequence of the three numbers of the "Ott"-valuation number follows their relevance. Most important are positive test results (in the hundredth position) followed by missing results of significant tests (in the tenth position). Less influence have negative test results (in the first position). From this a matrix of "Ott"-valuation numbers between 009 and 900, in respect to an increasing biochemical-toxicological potential, can be created (Fig. 6).



900									
810	801								
720	711	702							
630	621	612	603						
540	531	522	513	504					
450	441	432	423	414	405				
360	351	342	333	324	315	306			
270	261	252	243	234	225	216	207		
180	171	162	153	144	135	126	117	108	
090	081	072	063	054	045	036	027	018	009

Figure 6:
Matrix of "Ott"-valuation numbers

The total of the digits of the "Ott"-valuation number is always 9, so that for a rating, the first two numbers are sufficient. In addition, the negative test results are more heavily weighed. The more negative results are known the smaller the total of the digits of the first two numbers get. The tenth position contains the statement not or not sufficiently tested to a 50% probability of positive or negative classification after adequate testing. Therefore, the tenth position should be considered only by half its value when building the total of the digits. Exeptions are the values 018 and 009 which should go as "1" into further calculations to avoid multiplication by <1 or zero.

To combine "Tox"- and "Ott"-valuation numbers to one valuation number ("Tox/Ott"-valuation number) they are multiplied with each other. In panel "Tox" a valuation number of less than 11 is not to be expected, what would be comparable with a NOEL in picogram/kg. In other words, "Tox"-valuation numbers combined with "Ott"-valuation numbers have a spectrum from zero to 99 ("Tox" (max. 11) times "Ott" (max. 9) = 99).

The panel "Ca" (carcinogenicity) is with its five groups of classification on a higher level of relevance. Biochemical data are more normally considered as what is a consequence to the lack of "effect-orientated data" from groundwater contaminants. The carcinogenicity alters or substitutes for the values of the combined "Tox/Ott"-valuation number in the following way:

- Substances of group A and B_1 are, without considering their "Tox/Ott"-valuation number, classified always with "100".
- Substances of group B_2 receive in addition to their "Tox/Ott"-valuation number a "risk-rate" of +30, which is sufficient and does not alter the "Tox/Ott"-valuation number overproportionally. The total of 100 should not be exceeded. Therefore, a B_2 substance with a relatively high "Tox/Ott"-valuation number of 70 or higher is grouped equivalently to a A or B_1 substance.
- Substances of group D receive a "risk-rate" of +10 to their "Tox/Ott"-valuation number. This comparatively low addition is justified because group D will *also contain substances which are evaluated only by plausible relations of*

structure and effect (evaluation of substance classes in the case of missing data from individual substances).
- Substances of group C are evaluated only by the "Tox/Ott"-valuation number.

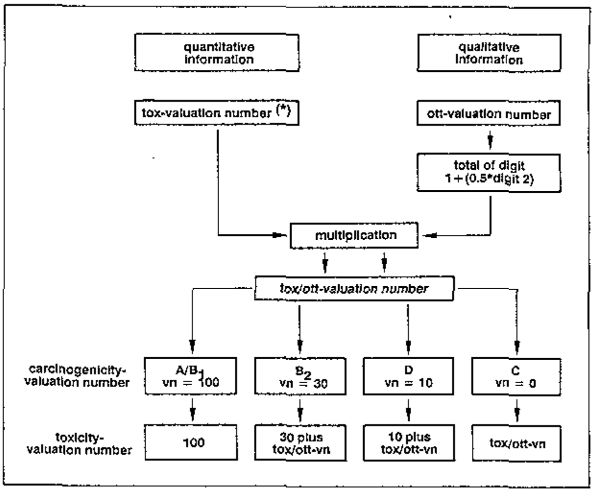


Figure 7:
Determination of the toxicity valuation number

With the described procedure (a summary is given in Fig. 7) the "toxicity potential" of a contaminant can be quantified (some examples are given in Tab. 1) so that it can be used for connection with other parameters.

The entire complex of the ecotoxicological evaluation cannot be dealt with in this space. Here, aspects of exposure are quite different to those discussed above. This means that specific evaluations for the ecosystem groundwater and surface water must be created. Ecotoxicological tests such as bacteria-test, algae-test, daphnia magna-test and fish-test should be considered.

	assessment-profile	tox/ott-vn	carcin.-vn	toxic.-valuation-number
dibutylphthalate	rel: III tox: 1.74 ott: 513 ca : C	9.6	0	9.6
tetrachloroethene	rel: III tox: 1.69 ott: 432 ca : B_2	9.3	+ 30	39.3
3,4-dichloro-aniline	rel: I tox: 6.62* ott: 270 ca : B_2	36.4	+ 30	66.4
benzene	rel: III tox: not ott: consid. ca : A	not consid.	+ 100	100.

Table 1:
Examples for the quantitative toxicological classification of different substances using the toxicological evaluation model

Integration of geochemical, physico-chemical and toxicological values

The standardized valuation numbers, standing for "the physico-chemical behaviour on the groundwater path", "the potential of toxicity" and for "the concentrations", have to be combined for an integrative valuation of a contaminant. One of several possibilities is the assumption of a linear increment of the substance-related hazard potential. In this case valuation numbers from zero to 300 would result. An other one is the usage of a threedimensional system of coordinates (Fig. 8). The following has to be filled in: On the z-axis the toxicity valuation number, on the y-axis the groundwater currency valuation number and on the x-axis the concentration valuation number. All three valuation numbers lie between zero and 100 so that a "valuation cube" with an edge length of 100 can be created (Fig. 8). A new "integrated valuation number" could then be received by using the ammount and the direction of a vector from the origin to a point inside the cube which is equivalent with a distinct groundwater hazard potential.

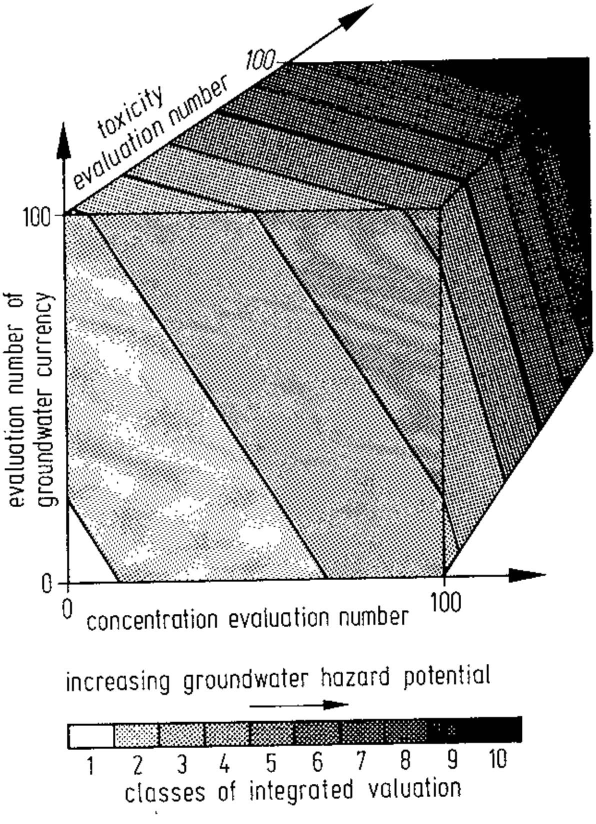


Figure 8:
Determination of classes of integrated valuation

In the case that they have to be weighed differently it would lead to a non-linear increment of the substance-related "hazard potential to groundwater" as it is shown in the "valuation cube" with a stressed toxicity valuation number (Fig. 8). This result of a standardized and integrated substance valuation is, for itself, of theoretical importance because only in connection with a special situation of utilization or exposure (e.g. drinking water wells in 200/500/1000m in distance of a contaminated site) does it become relative and leads to a realistic final assessment of a contamination source and the possible necessity of remedial actions.

The exposure assessment

An existing or planned utilization of contaminated groundwaters is an important aspect for the assessment of an exposure risk potential which possibly exists because of several contaminants. Very different kinds of utilizations and exposures exist which all have different demands for groundwater quality.

For drinking water purposes, other evaluation criteria have to be used than in water for industrial use and for the vulnerability and regeneration of the groundwater ecosystem (respectively surface water ecosystems) others than for the protection of valuable buildings. All of these have to be quantified to lead, in connection with the substance valuation, to a final commitment of remedial action priorities.

Integration of the parts "substance valuation" and "exposure assessment"

To date, all discussed parameters including the procedures for their integration are, due to a deficit of data, not yet fully applicable and should be regarded only as an aid for the estimation of risk potentials or sanitation priorities.

How a quantified and standardized "exposure assessment" in connection with the "substance valuation" is possible shall be explained by a drinking water production. The distance from the source of contamination, respectively the mass-streams of contaminants of the direct downgradient flow, and their parts of the water winning are to be considered.

A matrix is shown with decreasing portions of contaminated groundwater in drinkingwater production in the vertical direction, and the classes of the integrated substance valuation number (see Fig. 8) in the horizontal direction (Fig. 9). It may be used to classify remedial action priorities. Comparable to the "valuation cube", non-linear zones of a continuously increasing sanitation priority from black (immediate measures) to white (release into documentation) can be defined.

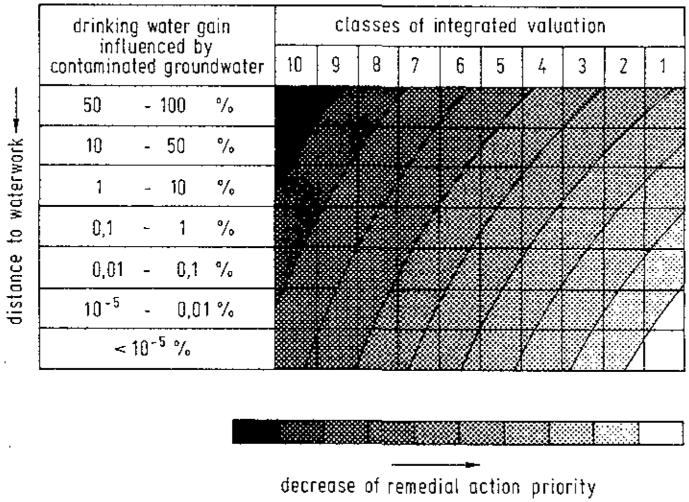


Figure 9:
Determination of remedial action priorities in the case of drinking water production downstream of a contaminant source

The shown example is hypothetical and only demonstrates the procedure of a non-linear "knotting" of "substance valuation" and "exposure assessment" to classify remedial action priorities. In a similar way other situations of usage or exposure can be combined with other requirements on groundwater quality to receive different kinds of instructions for activities. To achieve this purpose a great deal of research work has yet to follow.

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PHYSICAL, CHEMICAL AND DEGRADATION PROPERTIES OF CONTAMINANTS AT FORMER GAS WORKS

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INTRODUCTION

Redevelopment of gas works or other sites is complicated by the presence of by-products of industrial production that are almost always present at these sites.

In order to evaluate the potential effects of a chemical contamination on man and the environment, it is necessary to consider a number of different factors. These include the chemical composition of the contamination, its distribution in the environment, the routes by which it may reach target organisms and the potentially adverse effects on the target organisms.

Risk assessment is problematical in that there are many contaminants present, most of which are not characterised in the site investigation and each contaminant has several exposure pathways.

In this paper, we attempt to identify the hazards and routes of exposure in terms of the known composition and group properties of the actual contaminant mixture. Simple distribution and diffusion models using the physical-chemical properties of selected compounds representing the contaminant mixture are used to simulate behaviour patterns for the actual contaminants in the local environment around the sites of former gas works. Routes of exposure to man and the relevance of existing toxicological data can be assessed with a view to establishment of clean-up criteria.

CONTAMINANTS AT SITES OF FORMER GAS WORKS

In Denmark, there are approximately 112 gas works most of which are founded before or around the turn of the century. Generally the plants were initially rather small with horizontal retorts. Around the 1920's, many plants installed vertical retorts. Other improvements to production included changing the carbonisation temperature from about 700°C to over 1000°C.

Investigations at former gas works have shown that the most common form of contamination is due to spills of coal tar from underground tanks and waste pipes. In periods where the sale of coal tars was unprofitable there is a possibility that the coal tars and the sludges from the tar wells were disposed of on the site or at nearby waste sites.

Coal tars can contain as many as 10.000 components where only about 400 have been identified (IARC 1985). The characterisation of coal tars is further hamped by differences and difficulties with the analytical procedures. Generally the nitrogen and sulphur heterocycles are poorly determined. Due to the difference in carbonisation temperature and contact time in the retorts the composition of coal tars vary, differences occuring mainly in the relative proportions of the main components.

The main components in coal tars have been subdivided in the 7 groups shown in table 1.

		%
1.	Alkyl benzenes	2.36
2.	Tar acids	9.85
3.	Tar bases	1.77
4.	Other nitrogen (sulphur) & oxyen. heteronuclear compounds	2.11
5.	2 ring PAH compounds	9.10
6.	3 ring PAH compounds	7.44
7.	PAH compounds with 4 or more rings	54.54

Table 1.
Some constituents of coal tar, % by weight of crude tar for a coal from a horizontal retort with a temperature of 900-1000°C.
(Data from Coal tar Research Association through ref. Wilson & Stevens, 1981)

THE FATE OF COAL TARS IN THE SOIL ENVIRONMENT

In order to evaluate the effect of coal tar contamination on man, it is also necessary to estimate the distribution of the tar and its constituent components in the environment.

The crude coal tar is a viscous black liquid. The viscosity will depend on the composition of the tar i.e. the relative properties of the lower alkyl benzenes to the heavier pitch and the quantities of ammoniacal liquid emulsified in the tar (Verschueren and Visschers 1988). A crude coal tar with a relatively low viscosity can seep through the soil layers. Sandy soils will present very little resistance to this flow, whereas loams and peats can be effective in retaining the coal tars. The flow of coal tars in the more permeable soil zones can give rise to a horizontal movement and spreading of the contamination.

Often the coal tar can be observed as black streaks in the soils and in areas where concentration are high, the soil appears saturated. In areas where contamination is slight, the coal tar is presumably present as a thin layer on the surface of soil particles.

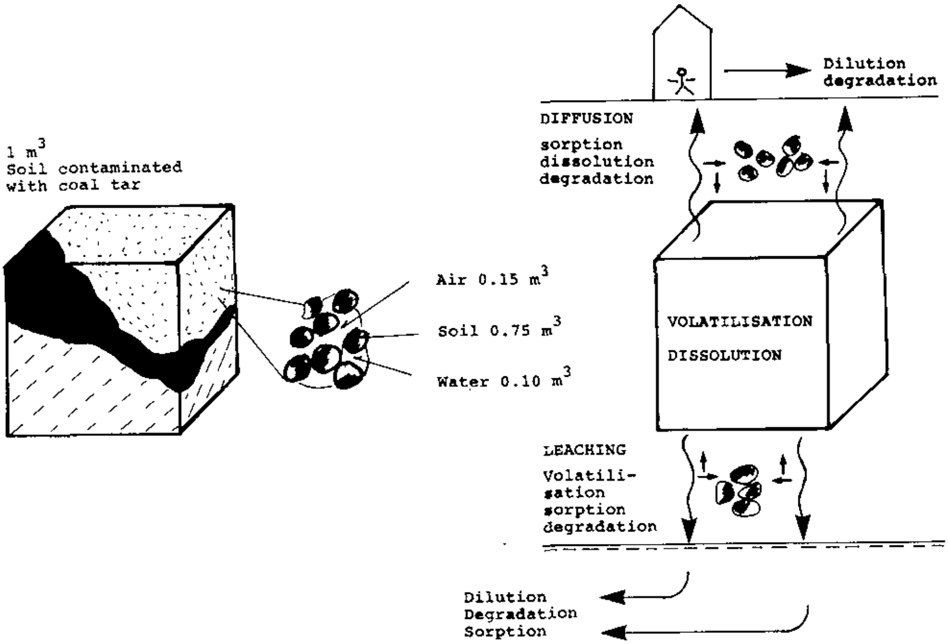
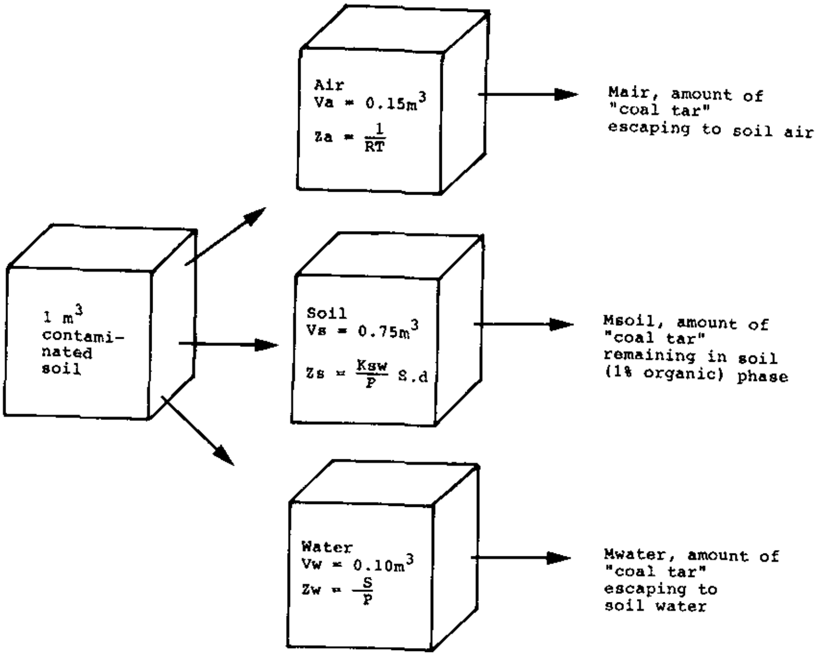


FIGURE 1

CONTAMINATION -> DISTRIBUTION -> TRANSFER

In figure 1, distribution to the three soil compartments, the soil particles, the soil water and the soil air, is illustrated. The distribution is dependent on the tendancy of the different coal tar components to escape from the mixture and partition into the various environmental compartments, especially the ground water and the atmosphere. These environmental compartments are large and equilibrium is not achieved due to their dynamic nature, whereby the hydraulic ground water flow and the wind will dilute and disperse the contaminants in the ground water and atmospheric compartments respectively. There are various models which attempt to describe at various levels of complexity the tendency of compounds to escape into the different environmental compartments and the reactions, physical, chemical and biological which occur in these compartments. One of the most useful models is that of Mackay (Mackay & Paterson, 1981), which in its simplest form is illustrated in figure 2.



R - Gas constant = 8.3143 J/K/mol

T - Kelvin

Ksw - Soil/water distribution = Koc. % organic carbon/100

Koc - organic carbon/water distribution

Log Koc = 0.524 log (Kow) + 0.855 (Briggs, 1981)

S - water solubility mol/m3 = (mg/l)/M

M - Molecular mass, g/mol

d - Soil density = 2

P - Partial vapour pressure, Pa ≈ vapour pressure x mass fraction

Total coal tar = 1 = VaZa + VsZs + VwZw

FIGURE 2

DISTRIBUTION BY FUGACITY APPROACH OF MACKAY & PATERSON (1981)

The essential determinants used in this model for calculatiing the distribution to the soil compartments are the vapour pressure of the pure substance, water solubility and the partition coefficient to the organic phase which is usually approximated by the octanol-water partition coefficient.

Models to describe environmental distribution agree reasonably well if only one compound is present. It is however difficult to predict the behaviour of such complex mixtures as coal tars. The calculation that follows has been carried out solely to obtain a crude estimate of the composition of tar components in each phase in order to relate this to known effects of coal tars after specific routes of exposure.

The distribution in the soil, soil water and soil air for each of the seven groups of coal tar contaminants (table 1) is calculated by assuming that one component in each group is representative for the other compounds. The seven compounds are benzene, phenol, quinoline, thiophene, naphthalene, acenaphthene and benzo(a)pyren. A typical distribution for different groups of coal tar compounds with different physical chemical properties is shown in figure 3.

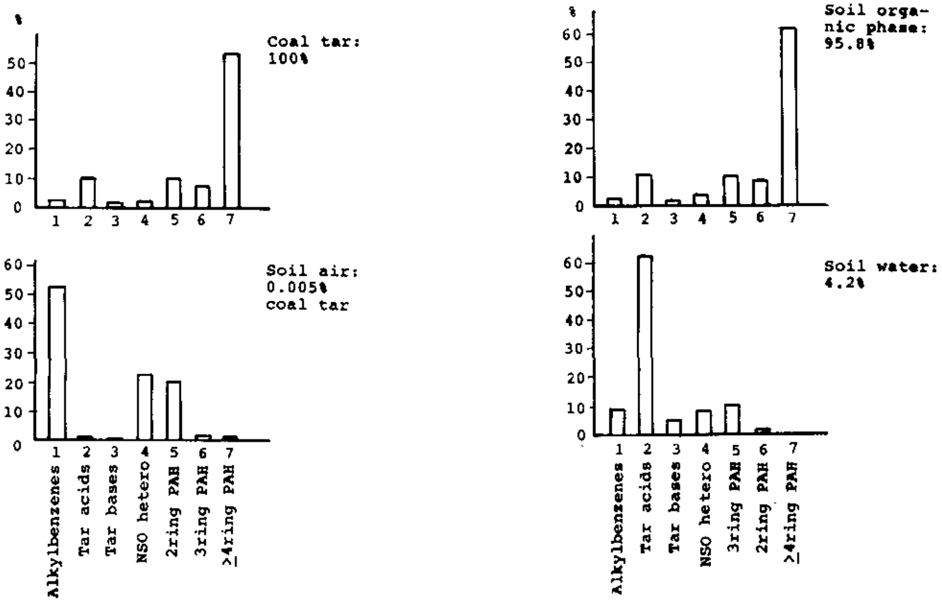
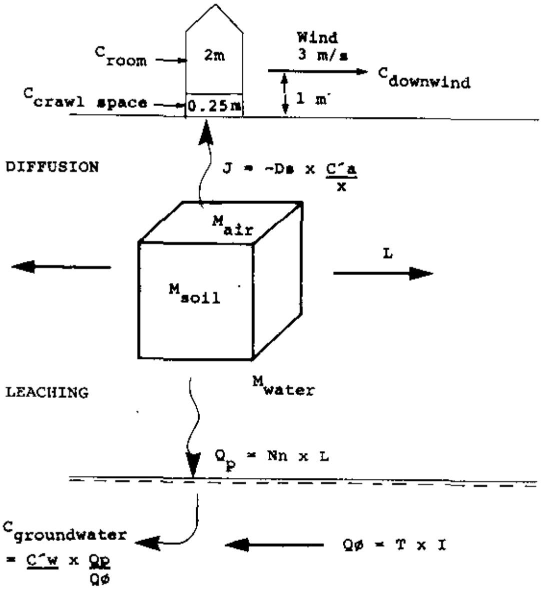


FIGURE 3

DISTRIBUTION PROFILES FOR COAL TAR & COAL TAR COMPONENTS IN THE SOIL ENVIRONMENT

TRANSFERS FROM THE CONTAMINATED SOIL LAYER

Volatilisation of contaminants to the atmosphere and dissolution of contaminants in the leaching rain water (figure 4) will remove some of the contaminants from the soil environment i.e. the coal tar profiles (figure 3) will change with time.



L - Length of contaminated area

Ds - Diffusion coefficient in soil $= Da \times (Va)^{2.20} \times 1.04$ (ten Berge 1986)

Da - Diffusion coefficient in air

x - Depth from soil surface to the contaminated subsoil surface

Ca - Concentration in air at the contaminated soil surface

Qp - Water flow from contaminated soil m3/year/m cross section

Nn - Precipitation m/year

Q0 - Water flow in aquifer

T - *Transmissivity m2/year*

I - Hydraulic gradient

C'w - Concentration in the leachate water

FIGURE 4

TRANSFERS FROM CONTAMINATED SOIL LAYER

Volatilisation

The flux or mass transport of each of the seven contaminant groups to the surface of the soil can be approximated by a simple diffusion model. It is assumed the contaminated soil is covered by a clean soil layer. The concentration in air at the contaminated subsoil surface is assumed to be that due to the mass of contaminant partitioning to air phase unless this exceeds the saturated partial vapour pressure.

This means that the diffusive flux for the individual heavier PAH compounds is independent of soil concentrations except at low soil concentrations.

Dissolution

The rain water will convectively remove the more soluble contaminants and the concentration in the ground water will depend on the dilution factor due to ground water flow. It is assumed that the concentration in the leachate water will approximate that due to the mass of the contaminant partitioning to the soil water unless this exceeds the water solubility. In practice, the concentration will almost always be less than the water solubility.

Mass transfer

Calculation of mass transfer by volatilisation and dissolution under different conditions shows that leaching can remove appreciable amounts of the alkyl benzenes, tar acids, tar bases, the NSO-heteronuclear compounds and the lower 2 ring PAH compounds within the course of 5 to 30 years if these components are fully "available". The coal tar profiles of both the soil and air and soil particles will be affected producing a relative higher proportion of heavier PAH compounds than shown in the profiles in figure 3.

If the site is covered (old foundations/-building) rain water will be negliable and vapour diffusion will control mass transfer. The elimination rate for even the most volatile component is relatively low if a covering soil layer is present and it is doubtful that appreciable mass transfer by volatilisation has occured since the cessation of gas production, (on average about 20-30 years).

It should be noted, that these more mobile coal tar components in the coal tar mixture might not be fully "available" in that a rate limiting step due to diffusion of the lighter components through the heavier pitch will control the flux of escaping components. Any physical disturbance of coal tar mass will create new surfaces and increase the flux (escape) of the more mobile components.

Biological degradation

The components escaping from the contaminated soil layer are available for biological degradation. Aerobic degradation will mainly occur in the ground water or in the water film on the surface of the soil particles in the unsaturated soil zone. The components remaining in the coal tar phase will not be available for biological degradation.

Almost all the components in coal tars can under favourable conditions be degraded albeit slowly (Kobayashi & Rittmann 1982). Of more importance is that many of the coal tar components e.g. the PAH compounds, will inhibit bacterial growth unless present in very low concentrations. The smaller substituted compounds are more amenable to bacterial degration degradation and these are also the more mobile in both water and air compartments.

Transport in groundwater

Once present in the groundwater the movement of contaminants away from coal tar source will occur although retardation due to partitioning with the organic soil phase will slow down the transport of the more hydrophobic components.

The diffusion of components in water is much slower than the diffusion in air by a factor 10^4, therefore the loss of volatile components from the surface of the water table in accordance with Henry's constant will be small in comparison to the concentrations in water, and unlikely to be of concern.

If the water table is very high and close to the soil surface there are possibilities of up-take in plants.

If the groundwater or leachate drains to a surface water catchment, there will be ecological consequences.

ROUTES OF EXPOSURE AND EFFECTS

The following discussion is deliberately limited to problems of direct human exposure that result from a future redevelopment, and ignores both indirect human exposure and effects on the environment in general. These effects would of course be included in a general evaluation prior to redevelopment.

Three routes of exposure are considered: inhalation, skin contact and ingestion.

Skin contact

As can be seen in figure 3, the coal tar present in the soil will have a very similar composition to fresh tar. In areas with low concentrations, there is the possibility that weathering (aging) will produce a tar with relatively higher proportions of the higher molecular weight PAH compounds.

Skin contact to coal tars will occur only where there is human access to contaminated surface soil. This may be the case if the redevelopment includes private gardens, childrens playgrounds, lawns or park areas. The length of time when potential exposure will occur will also vary, depending on the use. Preliminary estimates of daily dermal contact rates for children suggest figures of an average of about 1 and a maximum of 10 grams soil. Redevelopment schemes with paved or built up areas will in general present little or no possibilities for skin contact with contaminated soil.

Adverse effects of coal tar on human skin have been known for most of the present century. (IARC 1984, 1985). There are also many reports of coal tar producing skin tumours in animals after periodical application. Neither the human nor the animal data readily lends itself to setting reliable estimates of an acceptable lower concentration for coal tar in soil.

Coal tar can produce a range of effects other than cancer on human skin including cutaneous photosensitivity, patchy or generalised hyperpigmentation and hyperplasia. Patch testing demonstrates that many patients with eczema react positively to coal tar at low concentrations.

The uncertainties present in the estimates of daily exposure and of an acceptable exposure to a known carcinogen argue against attempting to set a fixed level for acceptable soil concentrations. Since a range of effects including cancer can occur at low soil concentrations, access to contaminated surface soil should be avoided in any redevelopment.

Inhalation

The profile of coal tar components in the air phase differs radically from the composition of crude coal tar.

However, certain qualitative estimates can be made. With age, the coal tar loses the more volatile fractions so that a relatively higher proportion of the heavier PAH fraction is present. The air profile has a tendency, with time, to resemble the original coal tar with exception that the tar acid and bases are always poorly represented.

Attempts to assess the carcinogenic potential of a mixture containing a variable composition of PAH on the basis of its individual components have in general not been very satisfactory. This is because there are usually other carcinogens in the mixture, and there are almost always a large number of other compounds present which act as co-carcinogens whose effect on the potency of the whole mixture it is not possible to predict.

There is human data for exposure to coal tar fumes on the basis of which estimates have been made for excess cancer risk. The US EPA has offered an upper bound lifetime cancer risk estimate of 62 per 100 000 exposed people per ug bènzene-soluble coke-oven emission per m^3 ambient air. Assuming a 0.71% content of benzo(a)pyrene in these emissions, it can be estimated that 9 out of 100 000 people exposed to 1 ng BaP per m3 over a lifetime would be at risk of developing cancer. (WHO, 1987). For benzene, WHO has estimated the lifetime risk of leukemia as 4 x 10^{-6} at an air concentration of 1 µg benzene per m^3.

For the PAH content, concentrations that could cause a lifetime risk of 10^{-6} or above based on the criteria quoted above will be below the normal analytical detection limits.

The theoretical models indicate that the diffusion of the heavier components is largely independent of soil concentrations. It can be shown that theoretical air concentrations in houses over contaminated soils containing a 100 mg "coal tar"/kg can be within the area of concern for both benzene and benzo(a)pyrene. The determining factors in actual situations will be the concentrations of the "available" contamination, the depth and nature of soil cover, the nature of the house's foundation and the ventillation in the crawl space under the house and in the house.

In conclusion, the building of houses over soils containing even small quantities of coal tar can present a theoretical risk of exposure to concentrations that could produce a life-time cancer risk of 10^6 or above, unless adequate measures are taken.

These measures will either require removal of essentially all contaminated soil that could potentially contribute to indoor exposure, or, more realistically, by ensuring adequate construction forms including ventilation to eliminate the possiblity of exposure.

Ingestion

Ingestion of coal tar components will be possible if there is access to surface soil with the opportunity for direct soil ingestion or the formation of contaminated dust. It may also occur if vegetables are contaminated or if drinking water supplies become contaminated.

People are not normally exposed to coal tars by ingestion, and there is very little data about the absorption or effects of these compounds following this route of exposure. However, precautions to prevent skin contact will also effectively prevent the possibility of exposure from contaminated surface soil and from vegetables. Criteria for limits of a variety of relevant coal tar components in drinking water have already been established and will not be discussed further here.

ACCEPTABLE FORMS OF REDEVELOPMENT

The evaluation given above has attempted to describe and quantitate elements necessary for setting limits for how much contamination is acceptable in soil. The difficulties in doing this are clear. It is difficult to characterise composition, to estimate exposure - both in terms of diffusion models for complex mixtures, and in terms of human daily exposure rate - and to predict safe levels particularly for known carcinogens. The analysis can however be used to indicate potential levels for concern for specific pathways and routes of exposure. This knowledge can then be used in the planning of a redevelopment in order to reduce human exposure by these routes as far as possible.

This practice of caution in respect of future development, should however not be interpreted as a statement that houses already built on contaminated sites necessarily present a threat to human health. These situations are far more complex than the problems discussed above. Remedial action in these cases will involve many other considerations than the planning of a new development and are beyond the scope of a simple risk analysis.

ACKNOWLEDGEMENT

This paper presents some of the preliminary results and conclusions of a project concer-

ning health and environmental problems associated with the redevelopment of gas works and similar sites. The aim of the project is to provide quidelines for future investigations and outline data requirements for hazard assessment of contaminated industrial sites. The project is sponsored by the Danish National Agency of Environmental Protection, however, the results and conclusions presented in this paper do not necessarily reflect the views of the Agency.

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THE USE OF MATHEMATICAL GROUNDWATER MODELS

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ABSTRACT

Mathematical models based on physical laws provide powerful tools for reliable analyses of groundwater systems. By utilizing the available data in a consistent manner parameter-distributed groundwater models cannot only scrutinize the current state of the systems; they are also capable of predicting the future states under unchanged or changed external conditions. The effects of planned operations of the system can effectively be evaluated by applying the model with input data representing the new conditions. This is possible, because the input data has a physical meaning, and any operation option can easily be translated into the mathematical model by appropriate adjustments in parameters, stresses and boundary conditions. By utilizing the predictive capabilities of a groundwater model the effects of alternative options can be evaluated, and hence provide the basis for management decisions.

Mathematical models can support the management of both groundwater resources and groundwater quality problems. Applications range from planning the development of groundwater resources including prediction of the effects of pumping wells to contamination problems related to point and diffuse pollutant sources.

The paper also emphasizes the data problem. Groundwater is not directly observable, and the geological environment in which the flow occurs can never be inferred in all details. Hence, a number of parameter values have to be estimated in a calibration procedure by sound hydrological and geological judgements. The model results are obviously no more accurate than the available data, and predictions will always include elements of uncertainty.

However, if the users of the model results recognize that a mathematical model of a groundwater system is an approximation to the complex conditions existing in the field, and that some portion of the parameter values are estimated, groundwater models represent valuable tools, which can support the management of almost any field problem.

1. INTRODUCTION

Groundwater is often referred to as the "hidden" component of the hydrological cycle (Konikow and Patten, 1985). It is not directly observable, and its existence and characteristics can only be inferred with some degree of uncertainty. But in many areas the groundwater resource is huge, and its occurrence and hydrological significance cannot be neglected in water management and planning.

In comparison with surface water the movement of groundwater is very slow; this implies higher residence times in the subsurface reservoirs and low flow rates. Also, the costs of withdrawing groundwater can be high particularly for deep and low-permeable formations. These circumstances have contributed to the fact that in most regions the surface water resources have been the primary focus of development. Only in a few countries like Denmark the groundwater resources are so abundant that the water supply is based entirely on subsurface water.

However, in response to the increased demands for water, the surface water resources have now become more fully developed particularly in arid areas where the surface water in general is less abundant. This has stimulated the interest in exploiting the groundwater resources and the conjunctive use of the two water bodies. Apart from the constraints which the uncertain knowledge of the groundwater system may impose on exploitation schemes, groundwater has some obvious advantages over surface water: better protection against pollution; little treatment required before diversion to user; constant temperature; small distances between source and user; and a fairly steady supply due to large storage capacities.

Development of groundwater systems implies pumping, and in order to design an appropriate pumping scheme the hydrological conditions must be investigated and interpreted. These interpretations form the basis for evaluating the pumping strategy under considerations to the recharge capability of the system and the possible adverse effects. The hydrogeological information is primarily obtained from measurements in pumping and observation wells. Since the number of observation points always is limited, the available fragmental information has to be interpolated in time and space in a consistent and reliable way. A mathematical model of the system is equipped to do this by compiling the available information and interpretations in a consistent way. The model enables quantitative predictions of the effects of alternative pumping schemes and can hereby help to improve the basis for management decisions.

In recent years groundwater management in many industrialized countries has also involved pollution problems. The economic progress in the late 50's stimulated the industrial production potential and also prepared the way towards high-consumption societies. Both factors led to a marked increase in waste products, both municipal and highly hazardous industrial wastes, which in most cases were disposed in unlined landfills. Usually the sites were covered by topsoil and cultivated; and being out of sight no problems were anticipated. Due to the slow movement of groundwater, contamination problems did not evolve for a long period of time.

However, within the last decade many countries have experienced a large number of contamination problems where leachate from the old waste-disposal sites unexpectedly has polluted large aquifer systems including wells for drinking water supply. In contrast to surface water, elimination of the contamination source has no immediate effect; contaminants already in the aquifer will continue to migrate and spread unless immobilized or removed. Similarly, pollution from diffuse sources such as scattered septic tanks and application of fertilizers and pesticides in agriculture has contributed to the deterioration of the groundwater quality.

Management of groundwater contamination problems involves an analysis of the existing conditions and prediction of the future migration of the contaminants through the groundwater system. Such analyses may show that remedial action is required in which case an appropriate method has to be identified and the effect assessed.

Contaminants are transported with the flowing groundwater, and hence the management of contamination problems is closely connected to the hydrogeological conditions. As mentioned previously these conditions are observed directly only at a limited number of points, and a consistent method of interpolation is hence required. Similarly, the delineation of the contaminant plume has to be carried out by interpolation between a set of observation points and the total amount of pollution expected to be present in the aquifer is obviously closely related to the pollution source, which often is difficult to quantify.

Again, mathematical models are useful tools for compiling all available information in a consistent manner. Application and calibration of a mathematical model provides a method for successive confrontations of hypotheses and interpretations of the system and available measurements. This procedure will ultimately result in a tool which integrates all available information and system interpretations in an optimal way. By simulating the transport of contaminants the model can give support to the assessment of the consequences of future spreading and hereby also to the decision whether remedial actions are required.

2. PRINCIPLES OF GROUNDWATER MODELS

All models including groundwater models are simplified representations of the real world. By simplifying reality the models can provide a good insight in the mechanisms and processes important to the problem being studied; yet, the models have to be complex enough to be representative of the system being studied.

Models can be either physical (e.g. laboratory sand tanks), electric analogue or mathematical. Since the latter group includes the most universally applicable and also the most widespread models, the present paper will entirely be devoted to this group.

A mathematical model consists of a set of equations which under certain assumptions describe the processes being studied. Mathematical models may further be divided into deterministic and stochastic types or combinations of these two. Here, only the deterministic type of models will be discussed; i.e. models which always produce the same output for a given input, other conditions being equal.

In general, deterministic models can be divided into two groups: (1) blackbox models which rely on purely empirically based relations between input and output; (2) physically based models which rely on the physical laws for the processes being considered.

Empirical models may be useful in applications where only analyses of past or present conditions are required. However, for groundwater problems, the benefit of using empirical models is small, and physically based models are always recommended as the reliability and consistency of the model results are much higher. This is particularly true when predictions are required for conditions that fall outside the historically observed range. In that case a physically sound description of the governing processes is a necessity. For contamination problems where both chemical and biological processes are involved one could similarly talk about chemically and biologically based models if they are based on theoretical laws. However, only physical processes will be discussed here.

In physically based models the processes are described by the governing partial differential equations for the individual processes. Such equations include the same parameters and variables that are used by hydrogeologists to describe the aquifer characteristics and behaviour. For groundwater flow models the important parameters are transmisivity, storage coefficient, and leakage coefficient, which all have the same interpretations in the model as in nature. Hence, model parameters and "real" aquifer parameters are interchangeable. Model simulations are carried out by solving the relevant differential equation for specified aquifer parameters, discharge and recharge and geologically defined boundary conditions. The outcome of the solution or the dependent variable, here the piezometric head, represents the current state of the system. This variable may vary in time as well as in space. The reliability of the simulations depends on how well the model represents the field conditions. Still, it has to be kept in mind that the field is very complicated and impossible to describe in all details, and hence simplifying assumptions are always required.

Solutions to the differential equations can be obtained by either analytical or numerical methods. Analytical solutions to the mathematical model are very restrictive, since they require the geological conditions to be homogeneous and the stresses uniform in time. More realistic descriptions can be obtained by solving the mathematical model on a computer using numerical techniques. Hereby, a larger variety of geological conditions can be included in the simulations. The present report will primarily be devoted to numerical models.

3. MATHEMATICAL FRAMEWORK

3.1 Groundwater flow

The flow of groundwater is described by Darcy's law, which states that the flow rate is proportional to the gradient of the hydraulic head, where the constant of proportionality is the hydraulic conductivity describing the hydraulic properties of the medium. The hydraulic conductivity varies in space in a manner which to a large degree is tied up with the spatial variation of the geological properties. Darcy's law is a well-proven relationship which has been used for quantitative assessments for over a hundred years. However, although simple in nature the applicability is often complicated by the large and unknown spatial variability of the hydraulic conductivity. Mathematically, Darcy's law can be formulated in tensor notation as:

$$q_i = -K_{ij} \frac{\partial h}{\partial x_j} \qquad i,j = 1,2,3 \qquad (3.1)$$

where q_i - specific discharge (LT^{-1})

K_{ij} - hydraulic conductivity (LT^{-1})

h - hydraulic head (L)

x_j - coordinate (L)

Combining the Darcy flow equation with the continuity equation which represents the conservation of fluid mass, the general partial differential flow equation is obtained (Fig. 3.1):

$$S_s \frac{\partial h}{\partial t} = \frac{\partial}{\partial x_i} \left(K_{ij} \frac{\partial h}{\partial x_j}\right) - W^* \qquad i,j = 1,2,3 \qquad (3.2)$$

where S_s - specific storage (L^{-1})

W^* - sink term (T^{-1})

t - time (T)

The equation is derived on the assumption that isothermal conditions prevail, and that fluid properties are homogeneous.

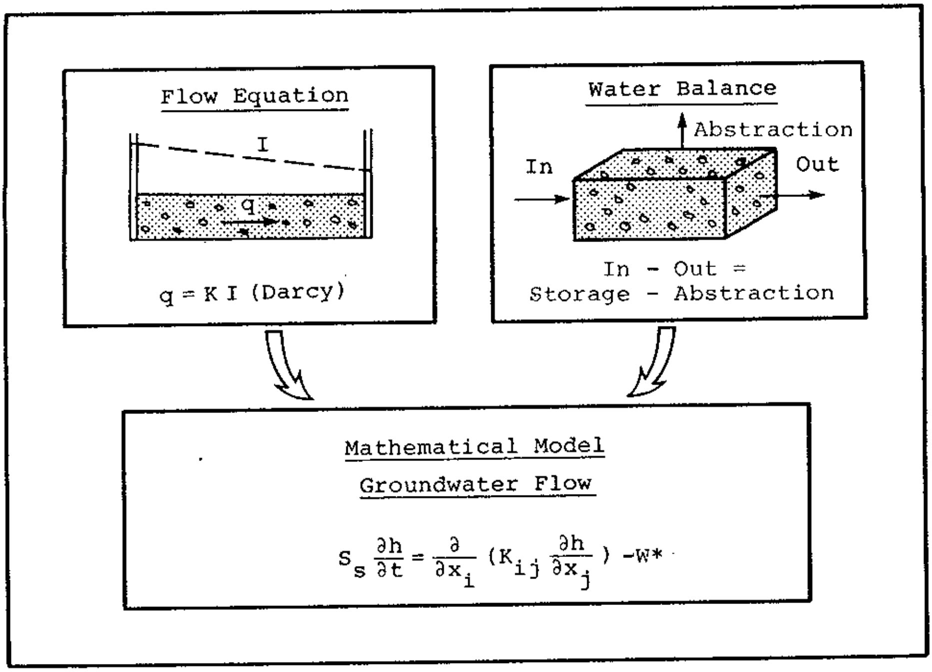

Fig. 3.1 Illustration of the derivation of the groundwater flow equation.

In most field applications a fully 3-dimensional equation (Eq. 3.2) is not warranted. Based on the assumption that the horizontal flow velocities are much more important than the vertical component the equation is often simplified to a two-dimensional areal flow representation by integration over the aquifer thickness b:

$$S \frac{\partial h}{\partial t} = \frac{\partial}{\partial x} (T_x \frac{\partial h}{\partial x}) + \frac{\partial}{\partial y} (T_y \frac{\partial h}{\partial y}) - W \qquad (3.3)$$

where S - storage coefficient ($S = S_s$ b)

T - transmissivity (T = K b) (L^2T^{-1})

W - sink term ($W = W^*$ b) (LT^{-1})

b - aquifer thickness (L)

x,y - horizontal coordinates (L)

All parameters and variables appearing in this equation represent the average aquifer properties over the saturated thickness.

The sink term W represents the external water exchange (volume flux per unit area), where W is positive for outflow and negative for inflow. The sink term may include: (1) well pumping or injection, (2) recharge, (3) steady leakage into or out of the aquifer through a confining separation layer to an over- or underlying aquifer and (4) water exchange with surface water through semipermeable river beds.

These external water exchange fluxes may conveniently be combined in a single expression (Konikow and Bredehoeft, 1976):

$$W = Q - \frac{K_z}{m} (H_s - h) \qquad (3.4)$$

where Q - rate of pumping (positive) or injection/recharge (negative) (LT^{-1})
K_z - vertical hydraulic conductivity of confining layer or riverbed (LT^{-1})
m - thickness of confining layer or riverbed (L)
H_s - hydraulic head in adjacent aquifer or river (L)

The simplified equation (Eq. 3.3) is more tractable because data requirements and available field data are in better agreement. Furthermore, the numerical treatment is less comprehensive than in three dimensions. The equation is strictly valid only for horizontal flow in confined aquifers, but it often also provides a useful approximation for horizontal flow in water table aquifers.

For a more detailed discussion of groundwater flow dynamics reference is made to Bear (1972), McWhorter and Sunada (1977) and Freeze and Cherry (1979).

3.2 Solute transport

Dissolved substances are transported in the aquifer by the flowing groundwater. The solutes may be either natural constituents or contaminents, and they are carried along the flow lines of the groundwater. However, due to the existence of the grain particles when viewing the process on a very small scale or due to the existence of the geologically based heterogeneities when a larger scale is adopted, the solutes are scattered and dispersed both longitudinally and transversely to the bulk flow direction. Traditionally, two transport components are identified:

1) Convection: transport related to the bulk flow field as derived from the available hydrological information.

2) Dispersion: transport related to flow components locally deviating from the bulk flow field.

In principle, also molecular diffusion contributes to the dispersion process. However, in real systems this process is usually of negligible significance.

The division into the above two components is somewhat obscure. In reality, transport of solutes is directly related to the velocity field of the flowing groundwater. However, in practice, it is impossible to derive the flow field which in all details represents the flow conditions in the aquifer, since a limited number of point measurements are available from which only the bulk motion of the groundwater can be described. Since the solutes are spread more than the bulk motion predicts, the concept of dispersion is introduced in which all other mechanisms conveniently are lumped. The dispersive effect is assumed to be linearly related to the bulk flow velocity through the dispersivity parameter.

This division into two interrelated components is a practical engineering approach of doubtful scientific relevance, and applications have clearly demonstrated that the dispersivity parameter has little versatility. If a detailed description of the flow field is established, the convective component accounts for a more dispersed migration of the solutes than if the flow field is described less accurately. Thus, in the first case relatively small dispersive effects have to be embodied in the dispersion term, and consequently a small value for the dispersivity parameter has to be adopted. On the contrary, for a less accurate water flow description an enhanced dispersivity value is required. These problems have stimulated the research within dispersion phenomena, and in recent years more interest has been directed towards stochastic approaches to the dispersion concept. However, such methods are still in the developing phase, and they are not fully operational yet. Hence, in the coming years solute transport predictions will still be based on the conventional convection-dispersion concept.

Following this concept, the flux of a dissolved species can be formulated in tensor notation as:

$$J_i = V_i C - D_{ij} \frac{\partial C}{\partial x_j} \qquad i,j = 1,2,3 \qquad (3.5)$$

Convection Dispersion

where J_i - mass flux of a dissolved species $(ML^{-2}T^{-1})$
V_i - seepage velocity (LT^{-1})
C - concentration (ML^{-3})
D_{ij} - dispersion coefficient (L^2T^{-1})
x_j - coordinate (L)

As it appears from the equation, the seepage velocity is the flow parameter which determines the convective transport. The seepage velocity can be obtained from Darcy's law:

$$V_i = - \frac{K_{ij}}{\varepsilon} \frac{\partial h}{\partial x_j} \qquad (3.6)$$

where ε is the effective porosity.

The dispersive transport is traditionally described by a Fickian expression where the transport is proportional to the concentration gradient with a proportionality factor (dispersion coefficient) being a function of the groundwater velocity and the properties of the geological formations.

Usually, the dispersion coefficient is described mathematically by the following equation (Scheidegger, 1961):

$$D_{ij} = \alpha_{ijmn} \frac{V_m V_n}{|V|} \qquad i,j,m,n = 1,2,3 \qquad (3.7)$$

where α_{ijmn} - dispersivity (L)
V_m - flow component in direction m (LT^{-1})
$|V|$ - magnitude of the velocity (LT^{-1})

As written above the dispersivity is a fourth-order tensor which includes 64 elements. However, for an isotropic aquifer the dispersivity tensor can be defined in terms of two constants (Bear, 1972): Longitudinal dispersivity α_L and transverse dispersivity α_T. These two parameters represent the property of the aquifer particularly the heterogeneity of the hydraulic conductivity.

However, as discussed above, the dispersion and also the dispersivity concepts do not provide a consistent representation of the physical processes causing dispersion. The values for the dispersivity parameters will be dependent on the scales at which the velocity and the concentration are defined. Hence, dispersivity values obtained in laboratory experiments are several orders of magnitude less than the values obtained in field investigations. A comparison of dispersivities obtained in the field is given by Lallemand-Barrès and Peaudecerf (1978), Fig. 3.2.

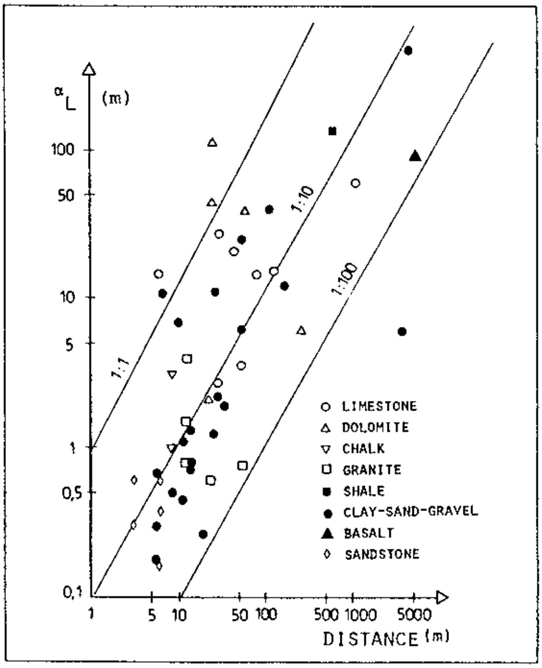

Fig. 3.2 Longitudinal dispersivity obtained by field tests shown as a function of transport distance. (Lallemand-Barrès and Peaudecerf, 1978).

The figure clearly indicates that the dispersivity increases with the transport distance. The larger spreading for larger transport distances can be interpreted as an integrated effect of the geological variability, because an increasing number of aquifer heterogeneities and hence velocity variations are encountered for larger aquifer scales or volumes. If the velocity variations are not included in the flow description, the spreading effect of these are compensated by an enhanced value of the dispersivity. The large unsystematic scattering of the dispersivity values (Fig. 3.2) resulting from the various field experiments hence represents the diversity in the scales at which the velocity is defined.

Besides the dispersive effect from aquifer heterogeneities, the seasonal variation in the groundwater flow pattern will also contribute to the spreading of solutes. Often this effect has also been compensated by increasing the dispersivity value thus contributing to the ambiguous pattern illustrated in Fig. 3.2.

More comprehensive discussions of dispersion concepts and problems can be found in Anderson (1979), Bear (1972), Gilham and Cherry (1982) and Neuman (1982).

Although obvious deficiencies are embodied in the dispersion concept, it represents a practical engineering approach which is adequate for many transport problems. However, care must be taken in transferring results from one field problem to another, and the influence of scale must be kept in mind.

Combining the transport equation (Eq. 3.5) with the mass conservation equation for the dissolved species, the partial differential equation for solute transport develops (the convection-dispersion equation), Fig. 3.3:

$$\varepsilon \frac{\partial C}{\partial t} = - \frac{\partial}{\partial x_i} (\varepsilon C V_i) + \frac{\partial}{\partial x_i} \left(\varepsilon D_{ij} \frac{\partial C}{\partial x_j}\right) - C' W^* \qquad i,j=1,2,3 \qquad (3.8)$$

where C' is the concentration of a dissolved chemical in a source or sink fluid (M/L^3). For reactive dissolved species additional terms would enter Eq. 3.8 to account for chemical and microbiological processes in the water or between the water and the aquifer material. Such reactions include decay, adsorption, ion-exchange, complexation etc.; however, in this document only physical processes will be considered, and no further discussions will be devoted to the reactions mentioned above.

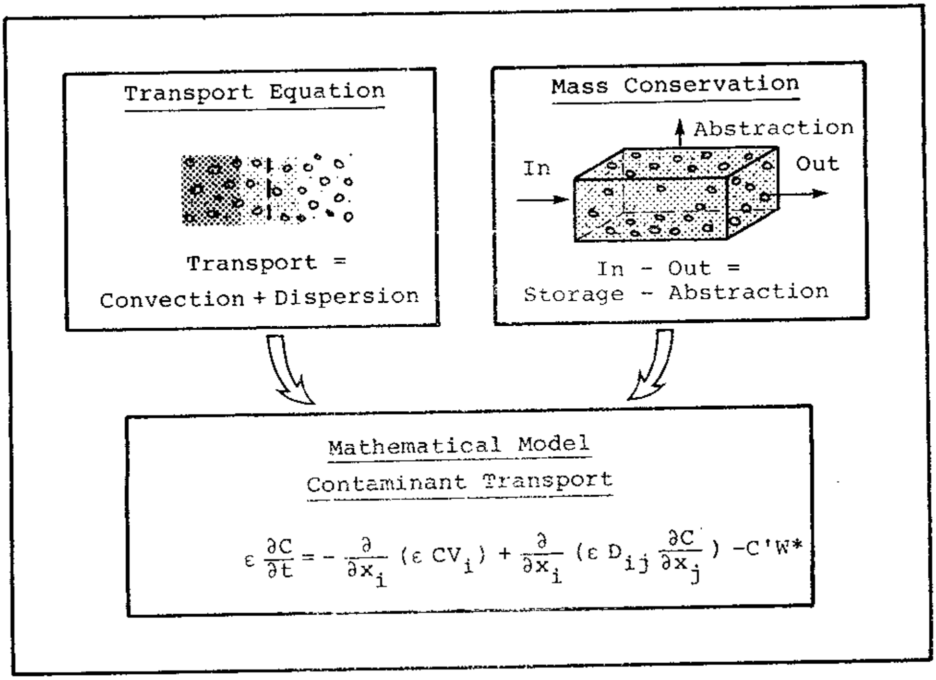

Fig. 3.3 Illustration of the derivation of the convection-dispersion equation.

In many practical applications one dimension of Eq. 3.8 can be eliminated. For regional contamination problems involving transport distances much longer than the saturated thickness, a two-dimensional areal representation may be adequate. Assuming that the concentration is uniform over the saturated thickness b, a vertical integration of Eq. 3.8 results in the following equation:

$$\varepsilon \frac{\partial}{\partial t}(Cb) = - \frac{\partial}{\partial x_i}(\varepsilon bCV_i) + \frac{\partial}{\partial x_i}(\varepsilon bD_{ij}\frac{\partial C}{\partial x_j}) - C'W \qquad i,j = 1,2 \qquad (3.9)$$

This equation suits the assumptions behind the two-dimensional transmissivity - based flow equation (Eq. 3.3).

Other contamination problems may conveniently be approximated by a two-dimensional cross-sectional representation. For example, when the aquifer thickness is comparable to the horizontal transport distance, the vertical dispersion cannot be neglected and consequently the areal representation is invalid. If the flow-lines within the contaminated region are parallel in the horizontal plane, a cross-sectional model may be applied to the field problem assuming that no significant flow occurs into or out of the plane of the section. In applying a cross-sectional model framework the parameters and variables represent the conditions per unit length perpendicular to the plane, and they are hence different from those in a horizontal model.

The two simplified model approaches described above are to some extent complementary and in many applications they can either alone or in combination provide a useful approximation to a fully three-dimensional description. This implies that data requirements as well as computational demands are reduced considerably. Many field problems will, however, still require a fully three-dimensional modelling approach.

4. SOLUTION TECHNIQUES

The partial differential equations discussed above represent the mathematical framework for analysing flow and contaminant transport problems. Since the equations are based on physically sound relationships which include parameters representing physical properties, boundaries and stresses of the system, solutions to the equations provide a mathematical description of the state of the system.

Both the groundwater flow equation and the convection-dispersion equation can be solved by analytical methods, if restrictive assumptions are imposed on aquifer properties and boundary conditions. The Theis-solution of the drawdown generated by a pumping well is a well-known example of an analytical solution which has experienced widespread applications. Although such solutions may be valuable for first-hand estimates, they are generally unable to recognize the true complex geometry of the system and the heterogeneity of aquifer properties, boundary conditions and stresses. Further, non-stationary conditions may make it impossible to obtain analytical solutions.

Numerical solutions to the flow and transport equations can effectively be obtained without the constraints imposed on analytical solutions. Although the numerical methods only offer an approximate solution, the numerical errors are usually insignificant compared to errors related to the conceptualization of the system and the uncertainty in the parameter estimation. In general, the possibility to include almost any information of the system in a numerical model more than outweigh numerical truncation errors.

Within groundwater hydrology either finite-difference or finite-element methods are generally used. These methods are currently being refined and made more effective, and as a result of the explosive development within computer technology still more comprehensive numerical models can be adopted by the computers. The present report will not review numerical methods in details, but reference is made to e.g. Remson et. al. (1971), Faust and Mercer (1980) and Wang and Anderson (1982).

Basically, all numerical methods involves a time and space discretization of the solution domain. The space discretization is introduced by dividing the region into a grid system, which may be rectangular or polygonal depending on the numerical procedure used. The solution to the governing differential equation is obtained by solving a set of algebraic equations over the selected grid system for specified parameters and boundary conditions. The time resolution is obtained by a consecutive updating of these algebraic equations which then are solved for each time step. An example of a vertical and horizontal grid overlay is shown in Fig. 4.1.

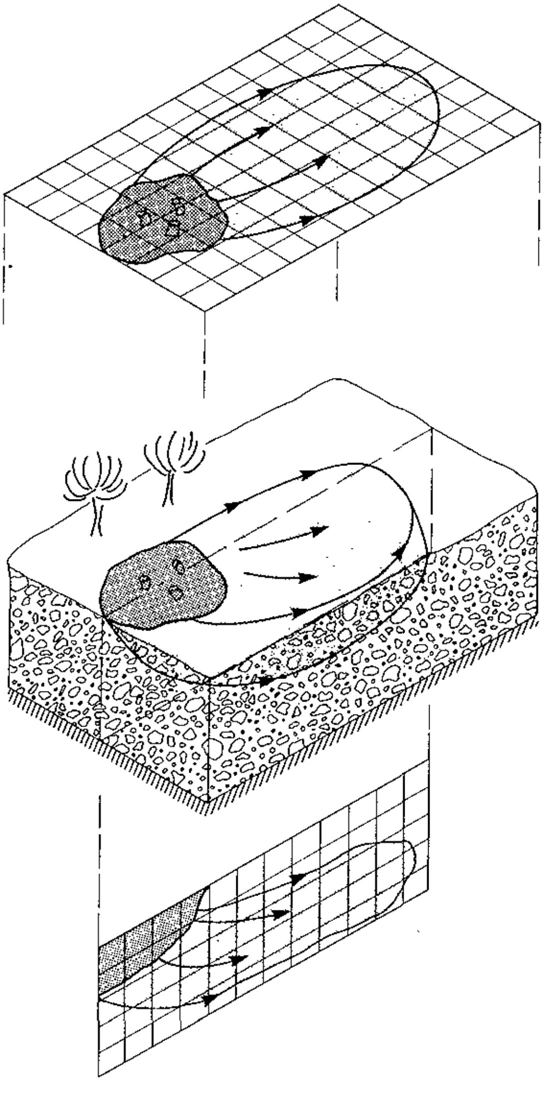

Fig. 4.1 Illustration of horizontal and vertical grid overlays to a field problem.

5. DATA REQUIREMENTS

Application of a numerical groundwater model requires a number of parameters to be specified within the model area. The following discussion of data requirements will be based on a two-dimensional horizontal flow and transport model applied to a confined aquifer, Fig. 5.1.

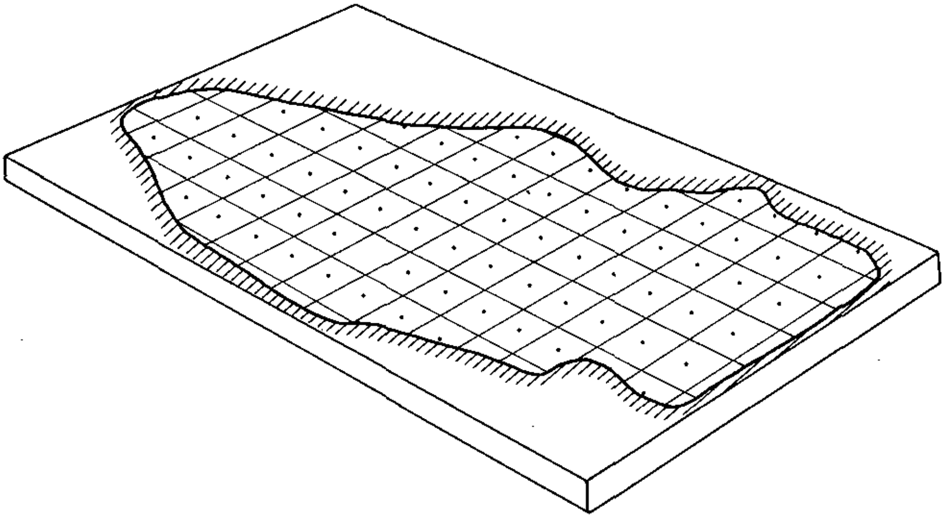

Fig. 5.1 Model grid of a two-dimensional horizontal groundwater model.

The size of the grid elements is selected with consideration to the objectives of the actual application. Smaller grid elements provide a more detailed description of the groundwater system, but on the other hand the requirements in terms of data specification and interpretation are larger and thus demanding a more elaborate effort of the model user. Further, the computational requirements increase proportionally to the refinement of the grid. All these requirements can be reduced by selecting a larger grid size which however will be at the expense of the model accuracy, since the grid elements are treated as homogeneous units. Hence, the selection of the appropriate space discretization is an overall compromise with consideration to the objectives of the study including the required accuracy of the model predictions, data availability and the available funding. For flow models typical grid sizes are from 25-100 m for local problems and up to 1000-2000 m for regional problems. Solute transport models generally operate on somewhat smaller grid elements.

Input requirements for a horizontal groundwater flow and transport model are listed in Table 5.1. Transport modelling can be interpreted as a superstructure to flow modelling, and such applications therefore require the specification of flow parameters as well as transport parameters.

The parameters, natural sources/sinks and boundary conditions characterize the aquifer system and can hence be inferred from the geological and hydrogeological information. Obviously, the geological conditions of the system may be more or less well-defined depending on the extent of field investigations. As a rule more than an exception the available information is always insufficient, and it is therefore up to the skill of the hydrologist to transform the real system into a mathematical model which preserves the essential elements of the system. The man-induced stresses on the aquifer such as pumping or leached pollutants from a landfill have to be assessed from available records. Initial conditions, which only are relevant for non-stationary simulations, specify the starting position for the simulation period. In general these conditions can be established from available records of hydraulic heads and concentrations. Based on the listed input specifications the model can simulate the aquifer behaviour in the form of predicted values of hydraulic head, flow and concentration as a function of space and time.

FLOW MODEL	TRANSPORT MODEL
Parameters • Transmissivity • Storage coefficient • Leakage coefficient • Porosity — Only required for subsequent transport modelling • Aquifer thickness All parameters to be specified at all grid elements.	Parameters • Longitudinal dispersivity • Tranverse dispersivity Compositional parameters representing the entire model area.
Sources and sinks • Pumping or injection through wells • Natural recharge or discharge To be specified where required.	Sources and Sinks • Concentration of dissolved species in leachate water from disposal site • Concentration of dissolved species in recharge or leakage water To be specified where required.
Boundary conditions • Impermeable barrier (no flow) • Prescribed flux • Prescribed hydraulic head Either option to be specified at model boundary elements.	Boundary conditions • Concentration of dissolved species in influent water at boundary elements To be specified at relevant model boundary elements.
Initial conditions • Piezometric level at start of model simulation To be specified at all grid elements for non-stationary simulations.	Initial conditions • Concentration of dissolved species at start of model simulation To be specified at all grid elements.
Predicted Variables • Piezometric head level • Flow velocity (Darcy) • Seepage velocity Predicted at all grid elements for each time step.	Predicted variable • Concentration of dissolved species Predicted at all grid elements for each time step.

Table 5.1 Input-output specification of a two-dimensional horizontal groundwater flow and solute transport model.

6. MODEL CALIBRATION

Application of a mathematical groundwater model for analyzing historical or predicting future conditions requires that the model accurately represents the processes in the aquifer.

A prerequisite is that the basic laws and assumptions on which the mathematical model is based suit the actual aquifer conditions it aims to represent. For example, if the flow model assumes a two-dimensional horizontal flow pattern, these flow conditions must prevail in the aquifer; otherwise the model results have little relevance. Similarly a two-dimensional cross-sectional application requires that no significant flow components prevail perpendicularly to the model section. All naturally occurring flow and transport processes in real aquifers are of three-dimensional nature, and hence the justification of a two-dimensional conceptualization must be carefully evaluated in each application.

If a model has been accepted as an appropriate representation for the aquifer system, an obvious requirement is that the model solves the underlying governing equation accurately. Applying numerical methods for solving these equations will always introduce numerical errors in consequence of the adopted discretization in space and time. In general, these errors can be reduced significantly by a proper discretization of the space and time dimensions, and, compared to other error sources of a model application, the numerical inaccuracies are of minor importance.

In most applications to field problems the quantitative assessment of aquifer properties usually is the predominant problem in tailoring the model to the system due to the insufficient knowledge of the geology. The parameters may only be quantitatively known in a small number of the grid cells, and the hydrologist is hence compelled to provide estimates for the remainder cells. Obviously, these circumstances highlight the necessity to demonstrate whether the simulation model is realistic or calling for a revision of the estimated parameter distribution.

This process, which is called "calibration", aims to minimize the difference between measured and predicted values of e.g. hydraulic head or concentration. The calibration is accomplished by adjusting the input data until the model reproduces the field conditions within an acceptable level of accuracy. The input data comprises, as indicated in Table 5.1, boundary and intial conditions; sources and sinks; and aquifer parameters. The distribution of the parameters is usually the least defined input data, and consequently the calibration of the model is often carried out by adjusting these until a proper simulation is obtained.

The most common calibration approach is a trial-and-error procedure in which the model response to successive adjustments in the input data is evaluated. This approach is highly subjective, and it requires a sound hydrological experience and judgement from the model user to effectively obtain a reliable calibration. The various parameters are interrelated, and the model output is an integrated response of the individual governing processes. In fact, the same output can be obtained by a number of different input configurations implying that a good calibration in itself is no warranty of a reliable prediction model. The reliability of the model may even be more warranted by the modeller's hydrological experience so that the physical structure of the aquifer system is not violated during the calibration phase, and values of the parameters are kept within proper ranges of variation. The calibration procedure may also give rise to a gradual revision of the geological interpretation of the system.

By including the modeller's hydrological experience and interpretaion of the aquifer system into the mathematical model to supplement the missing information of e.g. parameter distributions, the calibrated model provides an effective tool for analyses and predictions. The value of using a model is its ability to utilize and integrate all available data and information in a consistent way.

It must be recognized, however, that the model results are no more accurate than the available data. The model can not replace field data, but it can synthesize data. Hence, the predictions should be interpreted and used with due consideration to the degree of certainty by which the real system is represented.

7. EVALUATION AND SENSITIVITY ANALYSIS

Application of parameter-distributed groundwater models to field problems imposes data requirements which, as discussed above, in general are impossible to fulfil. The inadequate data basis necessitates that the parameter distributions and other input data have to be estimated partly by calibration. The calibration procedure may often result in a model description which compares well with observations. However, a good match in itself is not evidence of the validity or accuracy of the model, because many input configurations can result in the same model output.

For this reason it is important to evaluate the sensitivity of the model simulation to the parameters and variables which have been estimated by calibration. By changing the input data (boundary data, individual parameters, parameter distributions etc.) within likely ranges of uncertainty, the model reliability can be assessed by analysing the range of model outcome. By indicating the possible range of model predictions as dictated by the availability or rather shortage of field data, the decision-makers using the model results are provided with some indication of how much confidence they can place in the model. Further, sensitivity tests can also assist in setting up a priority list of additional data collection if the confidence in the model prediction is to be improved.

8. DEVELOPMENT AND USE OF MODELS

Models are usually developed within the scientific research world by synthesizing the research results into mathematical formulations. When the solutions of the mathematical equations are carried out on a computer, a numerical computer model evolves. In the initial phase of this process the model is often in a poor operational state, and only the developer himself can use it. Further inclusion of mathematical statements may widen the versatility of the model; however, such extensions are often accomplished by piecemeal programming, and the operational state may even become worse. By the time the model has been thoroughly tested and verified against measurements and has proved its applicability to field conditions, the computer program can be reorganized and a documentation written.

This process significantly enlarges the group of persons being able to apply the model and it can be transferred from the research world to consultants, decision makers etc.

The overwhelming development of computers enables a widespread application of the more comprehensive and sofisticated computer models. This is further supported by the latest trend in making the models "self-documenting" by including examples of typical parameter values and providing an interactive model execution. Although this development in computer capacity and programming style facilitates the applicability of models, the communication between model builders and model users should not be broken. An exchange of experience will definitely be of benefit to both parties.

A number of documented groundwater flow and transport models are available today covering the full range of 1-D, 2-D and 3-D conceptualizations. Basically they are all based on the mathematical framework discussed in Chapter 3, and the differences between e.g. 2-D horizontal models are mainly related to the numerical procedure adopted and the input/output facilities. Hence, from a professional point of view, it makes no major difference whether the one or the other is adopted. Often tradition and the specific interest of the model user determine the selection of the actual model.

For a comprehensive review of the available groundwater models reference is made to van der Heijde et al. (1985).

It should still be emphasized that useful applications of groundwater models, in particular to contamination problems, require a solid background in hydrology and geology. Also, despite the larger program support in model execution, a time investment is still required in order to become familiar with the model. These requirements to the user obviously increases with model complexity.

9. APPLICATIONS

Mathematical models provide powerful tools for groundwater management. By means of models the prevailing conditions of the system can be analyzed, and important processes of relevance for the problem can be quantified. Hereby it is possible to identify relevant management options. Perhaps the most important capability of physically based, parameter distributed groundwater models is the extrapolation beyond the existing conditions. The effects of planned operations of the system can efficiently be evaluated by applying the model with the input data being adjusted according to the new conditions. This is possible because the input data has a physical meaning, and any operation option can easily be translated into the mathematical model by adjustments of parameters, sources/sinks and boundary conditions. By utilizing the predictive capabilites of a simulation model the effects of alternative options can be evaluated and thus provide a solid basis for management decisions.

Possible applications of groundwater flow models include:

- Water balance studies
  - Assessment of regional patterns of inflow to the groundwater reservoir and outflow to surface waters, neighbouring reservoirs, wells etc.
- Well withdrawals
  - Prediction of effects of groundwater withdrawals on piezometric head levels and streamflow discharge.
  - Assessment of safe yield.
  - Prediction of movement of saline water interface
  - Prediction of effects of scattered groundwater withdrawal for e.g. irrigation.
- Changes in aquifer recharge
  - Prediction of effects of urbanization
  - Prediction of changes resulting from irrigation return flow and canal leakage
  - Analysis of longterm climatologically related trends in piezometric head levels and separation from man-induced changes.

- Parameter estimation
  - Assessment of regional distribution of the hydraulic parameters (inverse modelling).
- Planning of field investigations
  - Rationalization of data collection requirements by identifying the measurements most needed.
- Basis for transport modelling
  - Prediction of seepage velocities for subsequent use in transport modelling.
- Groundwater management
  - Optimal planning of the development of groundwater resources.
  - Seasonal subsurface storage of surface water.

Possible applications of groundwater quality models include:

- Contamination from point sources (e.g. waste disposal sites)
  - Assessment of existing contamination conditions.
  - Assessment of leachate concentration.
  - Prediction of future contamination conditions.
  - Identification of alternative remedial actions and predictions of the improvement on groundwater quality.
  - Identification of additional field investigations including appropriate locations of monitoring wells.
  - Site selection of future waste disposal sites and operational design.
- Contamination from diffuse sources (e.g. cultivated areas)
  - Assessment of existing groundwater quality.
  - Assessment of concentration in recharge water.
  - Prediction of future groundwater quality.
  - Identification of management options.
  - Prediction of effects of management options and of changes in cultivation practices.
  - Planning of monitoring network.
- Management of groundwater quality
  - Identification of pumping strategy to prevent saline water intrusion.

Various model case histories have been compiled by Jackson (1980) and Bredehoeft et al. (1982).

10. CONCLUSIONS

Management of groundwater, whether it concerns resource or contamination problems, involves a complicated set of processes and parameters interacting and related to each other. Since groundwater is not directly observable, and the geologic environment in which the groundwater is flowing is impossible to survey in all details, predictions of future conditions will inevitably include elements of uncertainty.

In addressing a groundwater management problem all conceivable information of relevance for the problem should therefore be considered. Deterministic groundwater models provide a method to synthesize the available information in a consistent manner, where aquifer parameters and variables are harmonized in accordance with theories of aquifer processes. Although the model assures consistency between the data and concepts, this in itself is not a complete warranty of the reliability of the predictions. In all field applications one always faces the problem of an inadequate data basis, which implies that a number of parameter values have to be estimated in a calibration procedure on the basis of general hydrological and geological interpretations of the system. For this reason it is important that the model calibration is combined with a solid background within these disciplines.

However, if the users of the model results recognize that an aquifer simulation model is an approximation to the complex conditions existing in the field and that parameter values often are estimated by calibration, deterministic groundwater simulation models represent valuable tools which can support the management of almost any field problem.

Models provide a means to analyse quantitatively the existing conditions and forecast the future behaviour of the system in response to imposed stresses. Furthermore, applying a model gives a better insight in the system behaviour, which may lead to revisions of initial interpretations. Support can also be obtained for establishing priorities for additional data collection. New information will always improve our understanding of the system and thus lead to revisions and improvements of the model. More reliable forecasts can consequently be issued.

Due to the uncertainty inherent in the outcome from model simulations, all applications should include a sensitivity analysis indicating the range of uncertainty of the predictions. Hereby, the decision makers have a better basis for evaluating the necessity of future investigations and to establish appropriate safety factors in design specifications.

Groundwater models experience a continuous development, and improved and more comprehensive models will become available. Future models will certainly be of a more interdisciplinary nature with elements from e.g. geology, hydrology, chemistry, biology and microbiology. Obviously, this imposes higher requirements on the professional background of the model users. However, mathematical models will when used disciplinarily allow for a better management of the groundwater resources.

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GROUNDWATER CONTAMINATION FROM AN INCINERATOR ASH AND HOUSEHOLD WASTE CO-DISPOSAL SITE

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Summary

A landfill site holding approximately 250,000 m^3 of municipal solid waste incinerator ash and household waste has caused a significant contamination of an adjacent aquifer. The groundwater contamination was detected by routine groundwater quality monitoring downstream from the site.

Subsequently, a comprehensive groundwater pollution study has been conducted in order to assess the health risks involved and to determine which remedial actions may be employed to prevent further contamination and to restore the groundwater quality in the area.

The activities have included installment of a number of multilevel groundwater sampling wells, frequent sampling and analysis of groundwater and leachate, performance of several test pumpings, laboratory investigation of migration characteristics of the leachate in the aquifer, and 3-D computer modelling of the water flow in the aquifer.

The contamination appears to be confined to upper part of the aquifer which consists primarily of sand with beds of coarser and finer materials. Eventually, it will spread to the lower part of the aquifer if no preventive measures are taken. Since the lower part of the aquifer which consists of chalk bedrock is being utilized downstream for extraction of drinking water remedial actions to prevent further spreading are recommended.

Several individual leachate plumes have been identified in the upper aquifer. Most of the plumes are characteristic of incinerator ash leachate but one plume has a high content of organic components which is usually associated with leachate from uncombusted garbage.

It is recommended that both short term remedial actions designed to stop the migration of the leachate plumes and long term remedial actions intended to reduce future emissions from the landfill site are implemented.

INTRODUCTION

Land disposal of residues from mass solid waste incinerators may pose a serious contamination risk to groundwater resources if the disposal site is located near a valuable aquifer. This is not necessarily due to leaching of compounds normally considered toxic but may also be caused merely by excessive leaching of soluble salts. Codisposal of incinerator residues with uncombusted domestic waste may complicate the disposal situation in terms of leachate properties and groundwater protection strategies.

This is illustrated by the case presented in this paper. A relatively young landfill containing incinerator ash and household garbage was identified as the source of a significant contamination of an adjacent aquifer. Some of the results of the ensuing groundwater pollution study and the principles of the recommended remedial actions are presented below.

SITE DESCRIPTION

The landfill

The landfill which is located at the site of a former gravel pit was filled up during the period 1979-1983. It has a 0.5 mm reinforced polyethylene bottom liner. The leachate is collected by a drainage system and conducted to a central sump from where it is withdrawn regularly by pumping. The site has a surface area of approximately 23000 m^2 and an average height of 10-12 m. Since 1983/84, the sloping surface has been covered by a 0.5 m layer of clay overlaid by a drainage layer and a layer of topsoil.

Figure 1 shows a plan view of the landfill site and the monitoring wells.

Estimates of the quantities of the various types of waste deposited are shown in **table 1**. The landfill was designed primarily for the disposal of incinerator residues and the household waste was placed at the landfill site only during periods of insufficient incineration capacity caused by accidents in 1979-1981. Although most of the household waste was placed in a separate (but not isolated) section of the landfill, the mixing of the highly organic household waste leachate with the incinerator ash leachate caused severe problems with formation of excessive concentrations of hydrogen sulfide at the early stages of operation of the landfill. This has been described elsewhere (Hjelmar, 1981).

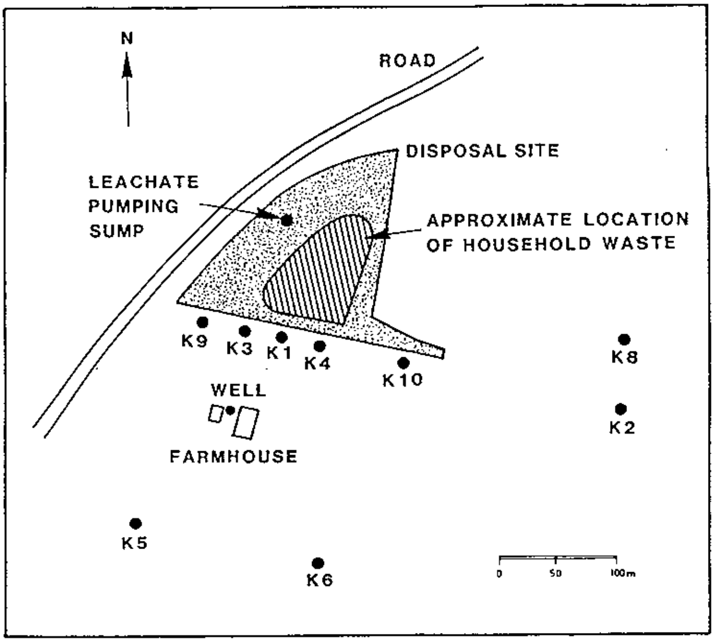

Figure 1 Plan view of the landfill site and the observation wells.

WASTE CATEGORY	QUANTITY DEPOSITED
Incinerator residues: Bottom ash, fly ash, and scrap metal	182000 m^3
Refuse from container site and recycling station	11000 m^3
Sludge from wastewater treatment plants	7000 m^3
Household waste	41000 m^3
Total	241000 m^3

Table 1 Approximate volumes of waste deposited at the landfill site.

The aquifer

The landscape surrounding the landfill is of glacial origin. The top layer consists of moraine of varying thickness (1-10 m). This is underlaid by 30-35 meters of fluvioglacial deposits consisting of a bed of fine sand with lenses or strata of both coarser and finer materials. The fine sand rests upon a layer of coarse gravel (0-5 m) which overlies the limestone bedrock (chalk from the Danian period).

As seen in figure 2, the water table is located in the upper part of the fluvioglacial deposits. In a regional perspective, the fluvioglacial deposits and the underlying chalk bedrock constitute one (semi-unconfined) aquifer. For the purpose of studying the migration of leachate plumes from the landfill, however, the local variations of the hydraulic conductivity of the various layers become important. Therefore, for practical reasons, the description of the hydrogeology refers to the aquifer as consisting of 3 horizontal layers, the A-, B-, and C-layers, respectively (see figure 2).

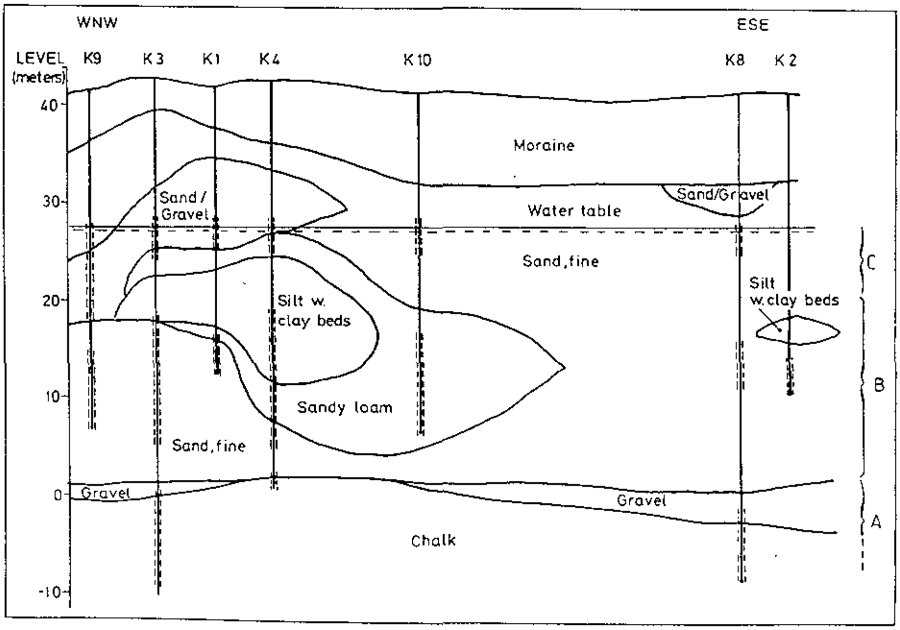

Figure 2 Geological cross-section at the south end of the landfill. The screens of the observation wells are indicated by broken lines. Separate casings were employed for each screen at each location.

The A-layer comprises the chalk bedrock and the layer of coarse gravel immediately above it.

The B-layer consists of the fluvioglacial fine sand with lenses of finer material and ends a few meters below the water table.

The remaining few meters between the B-layer and the water table constitute the C-layer of the aquifer. From the cross-section in **figure 2** it can be seen that the C-layer west of monitoring well K4 consists of sand/gravel whereas the C-layer becomes indistinguishable from the B-layer (fine sand) east of K4/K10. It is assumed that the highly permeable sand/gravel is also found underneath the landfill, the bottom of which is located at or just below the present water table. The extension of the sand/gravel layer to the southwest is not known.

The hydraulic properties of the various parts of the aquifer have been estimated from the results of physical examination of aquifer materials, pumping tests, observations of hydraulic heads, radioactive logging of boreholes, and modelling of the groundwater flow in the vicinity of the landfill site using a 3-D model (McDonald & Harbough, 1984). The transmissivity, T, of the A-layer was found to be approximately 2×10^{-3} m^2/s. The T-value of the B-layer was estimated at $1\text{-}7 \times 10^{-4}$ m^2/s. The thickness of the western part of the C-layer (sand/gravel) varies, and a permeability of $1\text{-}2 \times 10^{-3}$ m/s is estimated here. The permeability of the eastern part of the C-layer is approximately 8×10^{-5} m/s.

The general horisontal direction of groundwater flow in the area is south-southeast for both the A-, B-, and C-layer. In the C-layer, immediately south of the observation wells K9, K3, K1, and K4, a local, more southerly direction of flow is observed, due to the higher hydraulic conductivity of this part of the C-layer.

The general vertical direction of groundwater flow is downward from the C-layer to the A-layer. In the immediate vicinity of the landfill (e.g. at the observation wells K9, K3, K1, and K4), however, an upward gradient from the A-layer to the C-layer has been observed during 1987 and 1988. This is probably caused by the reduced groundwater recharge under the landfill.

The regional groundwater movements are governed by the annual withdrawal of some 2 mio. m^3 of groundwater from the aquifer (A-layer) at a public water supply facility approximately 3.5 km south-southeast of the landfill site.

LEACHATE

Quantity of leachate

During the period 1979-1987 approximately 27000 m^3 of leachate have been removed from the landfill site by tank car. The average annual amount of leachate removed corresponds to 350 mm (49% of the precipitation) during the disposal period and to 74 mm (12% of the precipitation) after the surface of the landfill was covered.

By comparing the amount of leachate removed from the landfill site with the amount of leachate produced at an adjacent incinerator ash monofill it was estimated that 1000-2500 m^3 of leachate may have leaked into the aquifer anually during 1980-1982. Since 1985, however, the annual amount of leachate leaking has been reduced to an estimated 300 m^3 due to the reduction of infiltration effected by the top cover.

Composition of leachate

Table 2 shows annual averages of the composition of the leachate withdrawn from the pumping sump. It is apparent that the initially high content of organics in the leachate caused by the presence of the household waste has decreased drastically since 1981, and the leachate is now typical of that produced at incinerator ash monofills (Hjelmar, 1987).

In 1986, a sample of leachate from the pumping sump was analyzed for a number of organic parameters. The results are shown in table 3.

In 1980, isolated pockets of leachate of much higher organic strength than the leachate collected at the pumping sump were located in the section of the landfill used for household waste. Percolation through the landfill was hindered by the layers of compacted moraine used to cover the daily deposits of household waste. Such pockets may still exist, and if they do, they are a potential source of groundwater contamination due to overflow of leachate over the side of the liner.

GROUNDWATER CONTAMINATION

Groundwater quality

The contamination of the aquifer was discovered in April, 1986 at the monitoring well K1B which at that time was the only observation well downstream of the landfill site. The only other monitoring well, K2B, was serving as a reference.

PARAMETER	UNIT	1979 - 1980 N = 6	1981 N = 5	1982 N = 4	1983 N = 4
pH	-	6.6 - 7.6	7.2 - 8.0	7.6 - 8.1	8.0 - 8.3
Conductivity	mS/m	3600	3300	2900	2900
Alkalinity	meqv/l	77	49	42	42
Chloride	mg/l	12400	10200	7400	8500
Sulfate	mg/l	890	1400	1500	810
Sulfide	mg/l	190	120	3.6	0.6
Total-N	mg/l	160	150	170	180
Total-P	mg/l	3.6	2.1	1.9	1.8
Boron	mg/l		4.4	2.8	4.7
Calcium	mg/l		310	130	72
Manganese	mg/l		63	38	48
Sodium	mg/l		4700	4900	4500
Potassium	mg/l		4200	4000	3600
Iron	mg/l	53	39	1.6	0.22
Zinc	mg/l	13	0.53	3.5	0.11
Lead	mg/l	0.20	<0.001-0.10	<0.005-0.49	<0.001-0.009
Cadmium	mg/l	0.0071	0.0030	0.0006	
Chromium	mg/l	0.030	0.014	0.007	0.007
BOD$_5$	mg O$_2$/l	1500	1300	38	27
COD	mg O$_2$/l	4600	2600	320	350

PARAMETER	UNIT	1984 N = 4	1985 N = 5	1986 N = 4	1987 N = 3
pH	-	8.4 - 8.7	7.1 - 8.5	7.5 - 7.9	7.5 - 7.7
Conductivity	mS/m	3100	3100	3380	3300
Alkalinity	meqv/l	32	43	54	55
Chloride	mg/l	9500	9600	9600	8900
Sulfate	mg/l	1700	680	590	680
Sulfide	mg/l	<0.05- 54	2.1	<0.5 - 0.72	<0.02- 0.10
Total-N	mg/l	220	220	260	300
Total-P	mg/l		2.9	2.6	2.7
Boron	mg/l				
Calcium	mg/l	36	39	150	110
Manganese	mg/l	16	25	44	63
Sodium	mg/l	4900	4700	5200	4200
Potassium	mg/l	3400	3500	3100	4600
Iron	mg/l	0.55	1.1	0.27- 7.3	1.0
Zinc	mg/l	0.10	0.22	0.54	0.23
Lead	mg/l	<0.008-0.016	0.017	<0.004-0.013	<0.006-0.017
Cadmium	mg/l	0.008	0.0007	<0.0004-0.002	0.0006
Chromium	mg/l	<0.003-0.008	<0.003-0.094	0.006	(0.011)
BOD$_5$	mg O$_2$/l	16	24	19	14
COD	mg O$_2$/l	340	370	370	350

Table 2 Annual averages of the composition of (unfiltered) leachate from the pumping sump. N = number of samples.

PARAMETER	CONCENTRATION IN LEACHATE	
NVOC	85	mg C/l
VOC	29	mg C/l
AOX	0.30	mg Cl/l
VOX	traces	
Phenol	0.0005	mg/l
Total phenols	0.021	mg/l
Chlorinated solvents	not detected	

Table 3 Analysis of leachate from the pumping sump sampled in October, 1986.

The well at the farmhouse (see **figure 1**) was also found to be contaminated in April, 1986. The well which drew water from the C-level was closed down and only used for monitoring purposes. An alternative water supply was provided.

At K1B, the level of contamination of the groundwater has increased constantly during the period April, 1986 - March, 1988 (**figure 3**). During the same period, no changes of groundwater quality were observed at K2B (**figure 4**).

Most of the water quality parameters shown in **figure 3** and **figure 4** are leachate pollution indicators. It may be noticed, however, that higher concentrations of calcium are found in the contaminated groundwater from K1B than in the leachate (**table 2**). This is often observed during the initial stages of groundwater contamination by leachate from incinerator residues and is probably caused by exchange of soil calcium with potassium, sodium, and ammonia which are usually present in high concentrations in the leachate (Kjeldsen et al., 1984).

Figure 5 and **figure 6** illustrate the changes of groundwater quality with time at two different observation wells which are both severely contaminated. Both observation wells (K3C and K4C) are located only a few meters from the south end of the landfill (see **figure 1**).

Figure 5 suggests a steady state situation at K3C, at least as far as the major pollutants (inorganic salts) are concerned.

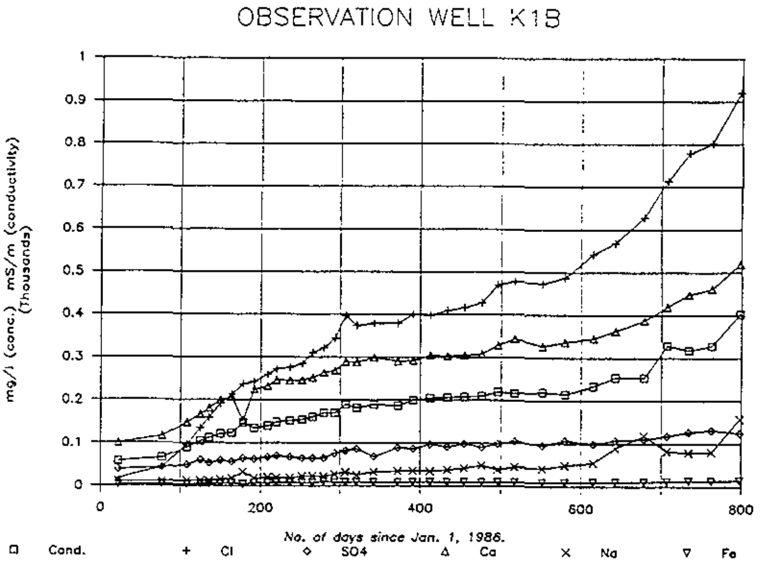

Figure 3 Groundwater quality parameters vs. time at monitoring well K1B.

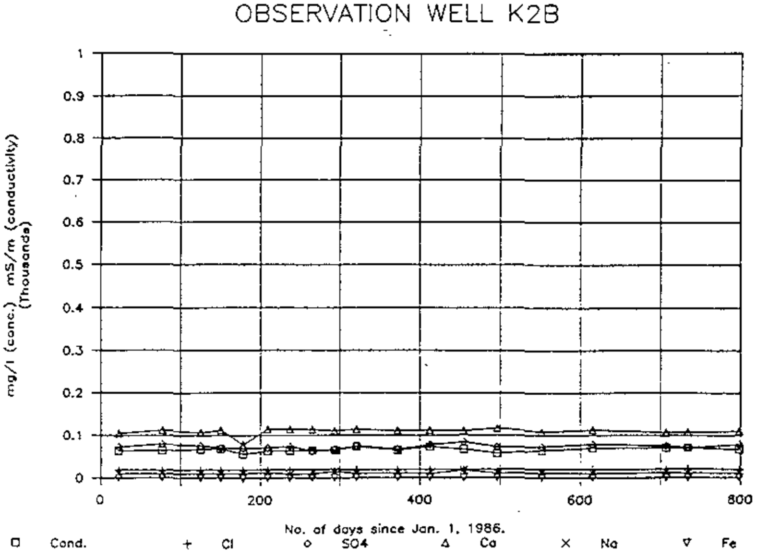

Figure 4 Groundwater quality parameters vs. time at monitoring well K2B.

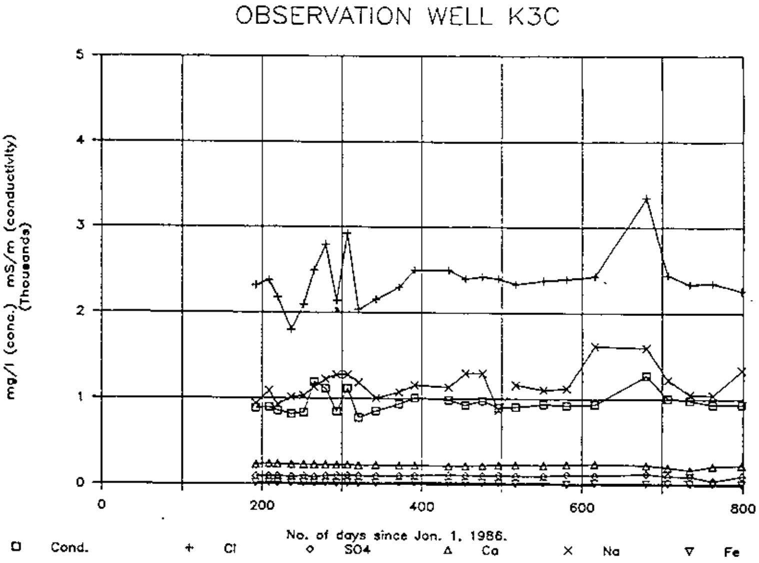

Figure 5 Groundwater quality parameters vs. time at observation well K3C.

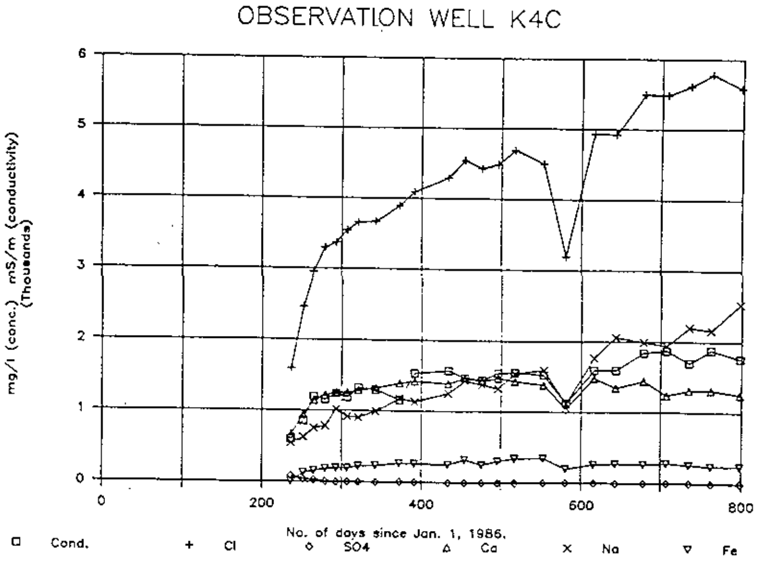

Figure 6 Groundwater quality parameters vs. time at observation well K4C.

The concentrations of potassium and ammonia are high at K3C (see table 4) and the calcium content is moderate. This indicates that the capacity for retention of potassium and ammonia and for exchange of soil calcium with leachate components has been exhausted in this part of the aquifer. The explanation may be that the contamination plume is 'old' at K3C and/or that the retention capacity of the coarse sand/gravel which constitutes the C-layer of the aquifer at K3 is low.

The groundwater quality at K4C exhibits a more transient behavior. The contamination level is constantly rising, potassium and ammonia concentrations are still low, and calcium concentrations are high, which are the characteristics of a 'younger' plume. The most striking dissimilarity, however, between the groundwater quality at K3C and K4C is the difference in the content of organic pollutants. This is illustrated in table 4 and table 6. The high concentration of organic components in the groundwater at K4C must reflect a leakage of leachate into the aquifer from that part of the landfill in which household waste predominates. The reducing nature of the organic plume has caused the concentration of sulfate at K4C to fall from an initial value of approximately 80 mg/l to less than 2 mg/l.

Apart from trace amounts of chlorinated solvents, no significant organic contamination was found at any of the other observation wells. The concentration of non-volatile organic carbon (NVOC) in the leachate from the pumping sump was 4-8 times lower than that observed at K4C.

Concentrations of cadmium, chromium, copper, and lead in the groundwater did not deviate significantly from natural background values.

The transfer of heat from the landfill to the aquifer is sufficiently high to elevate the temperature of the groundwater several degrees at the closest observation wells.

Plume origin and extension

The data suggests that leachate is leaking from more than one location within the landfill into the aquifer. The multiple point sources of leachate of varying composition contribute to the formation of several contamination plumes some of which may overlap each other.

PARAMETER	UNIT	K1C	K3C	K4C	K5C	K6C
pH	-	6.8	6.9	6.5	7.2	7.3
Conductivity	mS/m	162	940	1770	82	66
Alkalinity	meqv/l	7	14	14	4.8	5.2
Chloride	mg/l	180	2260	5590	42	11
Sulfate	mg/l	34	110	2	54	56
Nitrate	mg/l	0.03	2.2	<0.01	35	3.1
Ammonia	mg/l	0.68	41	0.28	<0.01	0.08
Calcium	mg/l	210	235	1260	131	112
Sodium	mg/l	21	1340	2540	16	12
Potassium	mg/l	5	964	5.5	1.5	
Iron	mg/l	21	17	250	0.27	0.28
NVOC	mgC/l	7.7		360	9.8	2.3
Boron	mg/l	0.3				
1,1,1-trichloroethane	mg/l	0.0005	nd	0.00003	0.00026	nd
Trichloroethylene	mg/l	0.0002	nd	0.0029	nd	nd
Tetrachloroethylene	mg/l	0.00007	nd	0.00006	nd	nd
Temperature	°C		15	13	8	9

nd: not detected

PARAMETER	UNIT	K8C	K9C	K10C	FARM WELL
pH	-	7.2	7.2	7.2	7.0
Conductivity	mS/m	78	66	800	264
Alkalinity	meqv/l	4.7	4.1	7.3	8.1
Chloride	mg/l	25	27	2180	345
Sulfate	mg/l	78	91	198	65
Nitrate	mg/l	1.5	3.3	7.8	
Ammonia	mg/l	0.04	0.11	0.02	3.8
Calcium	mg/l	114	110	980	168
Sodium	mg/l	16	10	440	265
Potassium	mg/l	5.9	2.4	8.3	117
Iron	mg/l	0.55	0.50	0.85	0.22
NVOC	mgC/l	1.9	1.8	2.2	2.6
Boron	mg/l				0.3
1,1,1-trichloroethane	mg/l	nd	nd	0.00004	0.0002
Trichloroethylene	mg/l	0.0001	nd	0.00008	nd
Tetrachloroethylene	mg/l	nd	nd*	nd**	nd
Temperature	°C		9	9	10

nd: not detected

*: chloroform: 0.0002 mg/l **: chloroform: 0.0010 mg/l

Table 4 Analyses of groundwater from the C-layer (March, 1988).

PARAMETER	UNIT	K1B	K2B	K3B	K4B	K5B
pH	-	6.8	7.2	7.2	7.1	7.1
Conductivity	mS/m	403	68	52	60	93
Alkalinity	meqv/l	4.8	4.7	4.2	4.1	5.2
Chloride	mg/l	921	21	14	19	69
Sulfate	mg/l	126	80	42	50	66
Nitrate	mg/l	<0.01	<0.01	<0.01	<0.01	21
Ammonia	mg/l	0.44	0.02	0.15	0.13	0.02
Calcium	mg/l	520	112	85	92	152
Sodium	mg/l	159	12	11	13	21
Potassium	mg/l	5.4	2.8	2.2	2.2 2.2	
Iron	mg/l	15	1.1	2.1	2.7	
NVOC	mgC/l	2.7	2.5	1.3	1.9	1.6
Boron	mg/l	0.4				
1,1,1-trichloroethane	mg/l	nd	nd	nd	nd	0.00006
Trichloroethylene	mg/l	nd	nd	nd	nd	nd
Tetrachloroethylene	mg/l	nd	nd	nd	nd	nd
Temperature	°C	12	8	10	10	8

nd: not detected

PARAMETER	UNIT	K6B	K8B	K9B	K10B	K3A
pH	-	7.3	7.5	7.2	6.8	7.3
Conductivity	mS/m	62	54	50	60	52
Alkalinity	meqv/l	4.3	3.6	4.2	4.0	4.4
Chloride	mg/l	23	17	15	21	12
Sulfate	mg/l	43	71	32	63	42
Nitrate	mg/l	2.2	0.59	<0.01	<0.01	<0.01
Ammonia	mg/l	0.08	0.04	0.16	0.09	0.22
Calcium	mg/l	101	90	83	98	83
Sodium	mg/l	10	9	10	11	11
Potassium	mg/l	2.2	1.9	2.0	2.5 2.1	
Iron	mg/l	0.49	0.71	2.0	0.16	2.2
NVOC	mgC/l	2.7	1.7	2.7	1.4	2.5
Boron	mg/l					
1,1,1-trichloroethane	mg/l	nd	nd	nd	nd	nd
Trichloroethylene	mg/l	nd	nd	nd	nd	nd
Tetrachloroethylene	mg/l	nd	nd	nd	nd	nd
Temperature	°C	8	8	9	9	9

nd: not detected

Table 5 Analyses of groundwater from the B-layer and K3A (March, 1988).

PARAMETER	UNIT	K3C	K4C
NVOC	mg C/l	13	340
VOC	mg C/l	<0.5	7
AOX	mg Cl/l	0.1	0.12
VOX	mg Cl/l	<0.04	0.23
Chloroform	*mg/l*	*<0.0002*	*0.0021*
1,1,1-trichloroethane	mg/l	0.0003	0.0032
Trichloroethylene	mg/l	0.0002	0.0075
Tetrachloroethylene	mg/l	<0.00003	0.0003
Phenol	mg/l	0.001	<0.0001
Phenols (total)	mg/l	0.028	0.46

Table 6 Analyses of organics in contaminated groundwater from observation wells K3C and K4C (September, 1986).

The plume pattern presented in figure 7 is consistent with most of the available information. A plume is here defined as that part of an aquifer in which the quality of the groundwater differs significantly from the background condition due to migration of leachate.

Plumes 1, 2, and 3 are located primarily in the upper part of the *aquifer (the C-layer)*. At a distance of 50-100 m from the landfill the plumes begin to move downwards and they may reach the A-layer in 10-20 years.

Plume 1 is contaminated with incinerator ash leachate, and the leak responsible may be located at the leachate pumping sump. Plume 2 is contaminated with leachate which is strongly influenced by household waste. The leachate forming plume 2 may be spilling over the side of the liner at the south end of the landfill section in which household waste predominates. Plume 3 is contaminated with incinerator ash leachate and may originate at northern end of the landfill, possible due to some failure of the bottom liner.

Plume 4 which is located in the B-layer is probably moving upward. The plume is contaminated with incinerator ash leachate which may have leaked out, for instance at the pumping sump, and moved downward some years ago under different hydraulic conditions.

No contamination of the B-layer was detected at K9, K3, K4, and K10. No contamination of the A-layer was detected (see table 5).

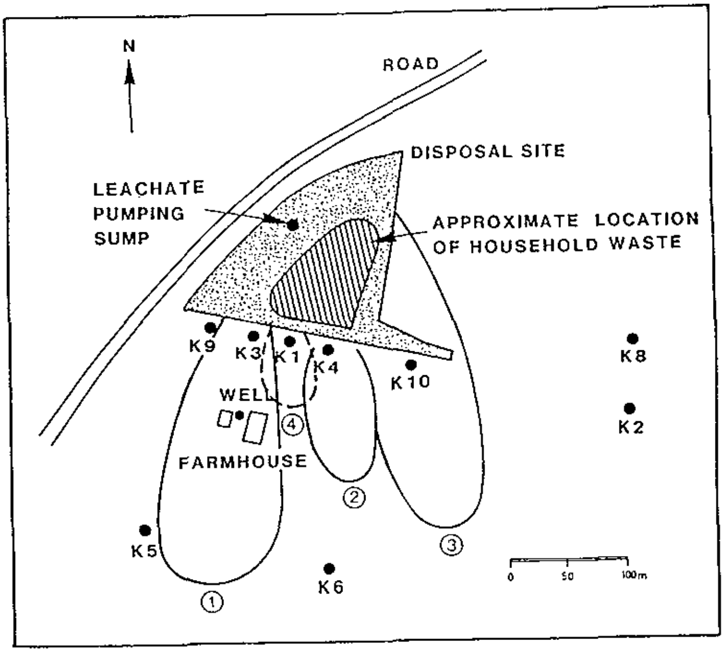

Figure 7 The estimated horizontal extension of the leachate plumes in March, 1988.

PRINCIPLES OF REMEDIAL ACTIONS

Risk assessment

A risk assessment has indicated that the landfill does not appear to pose an immediate hazard to downstream drinking water interests. Over a longer period of time, however, the risk of adverse effects of the groundwater contamination on downstream drinking water wells drawing water from the chalk bedrock cannot be ignored. It was therefore recommended that steps be taken to prevent further migration of contaminated groundwater in the aquifer and to reduce future emissions of leachate from the landfill.

Short term remedial actions

Test pumpings and model simulations indicate that selective pumping of a total of 8-10 m^3/hr of contaminated groundwater from 2-4 downstream wells will be sufficient to stop the migration of pollutants and eventually to rinse the more contaminated parts of the aquifer. The pumping strategy will intend to minimize the amount of water that must undergo treatment prior to discharge into a surface water body.

Long term remedial actions

The pumping scheme outlined above will stop the groundwater pollution from spreading and it will clean up the aquifer. But it will not influence the source of the contamination itself.

It does not seem practical or economically attractive to extend the period during which the pollution is controlled by pumping of groundwater beyond maybe 5-20 years. However, the incinerator ash in the landfill may be expected to produce leachate of considerable strength for a period of maybe 100 years or more. Therefore, the short term pumping of groundwater should be supplemented by remedial measures designed to minimize the long term rates of transfer of pollutants from the landfill to the aquifer.

The ultimate goal is a landfill site which can be abandoned safely within a manageable future (e.g. 20-30 years) because the rates of transfer of pollutants to the surroundings have become low enough to be of no environmental concern.

This might be achieved either by permanently reducing the rate of production of leachate in the landfill (e.g. by improving the long term efficiency of the top cover and draining system) or by irreversibly reducing the concentration of pollutants in the leachate to a low level (e.g. by enhancing the initial rate of leaching and/or degradation by temporarily increasing the rate of infiltration and percolation through the landfill and removing and discharging/treating the leachate). A combined solution comprising enhanced leaching followed by reduced production of leachate might be favourable.

CONCLUSIONS

Some of the conclusions that may be drawn from the study are:

The contamination appears to be confined almost entirely to the upper part of the aquifer. It does not constitute an immediate hazard to downstream drinking water resources.

Eventually, the contamination will spread to the lower, more valuable aquifer. Therefore, remedial actions should be implemented.

Both short term remedial actions designed to stop the migration of the leachate plumes and rinse the aquifer and long term remedial actions intended to reduce future emissions from the landfill should be carried into effect.

The dual nature of the leachate(s) - inorganic ash leachate and organic garbage leachate- may somewhat complicate both the short and long term efforts to achieve the ultimate goal: a landfill which can be abandoned safely within a manageable future.

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ON-SITE LANDFILL LEACHATE TREATMENT BY LONG-TERM AERATION AND INFILTRATION - AN EXAMPLE FROM MÅSALYCKE, SOUTH SWEDEN

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Summary

On-site leachate treatment as utilized at Måsalycke landfill consists of following steps: leachate collection, storage and flow equalization, long-term aeration, sand filtration and infiltration via open ditches. Data for 1987 shows high treatment efficiency for both organic, nitrogen and metallic compounds. The brook draining the area is partly affected by the inputs of solutes from the infiltration area. However, mean increase of concentrations in the brook is only about 5% for total salts and much less for other constituents. Particularly effective is the removal of nitrogen compounds by the nitrification (in the aeration step) and denitrification processes (during infiltration of pre-treated leachate).

INTRODUCTION

In Sweden, leachate from large municipal landfills is usually treated in conventional wastewater treatment plants, together with municipal waste water. This practice is often costly, especially if leachate is to be transported by special pipings or trucks for longer distances. In addition, wastewater treatment process may be disturbed due to high salt or metal concentrations in the leachate, and low-degradable organic compounds often pass through conventional treatment facilities without any decomposition. Finally, large quantities of cold leachate which are produced during snowmelt periods or rainy autumns, frequently lead to serious disturbances in wastewater treatment process.

In order to avoid these problems, a system of on-site leachate treatment has been developed and tested at Måsalycke, South Sweden, and is shortly described herein. Further details may be found in Nilsson & Persson (1984) and Nilsson & Vanek (1987 and 1988).

SITE DESCRIPTION

Måsalycke is a municipal landfill localized in south-eastern Scania (Figure 1) and serving Simrishamn and other nearby located communities since 1975. Annual disposal rate is about 30.000 tons of mixed wastes and landfilling area covers ca. 8 ha. On annual basis, precipitation exceeds evaporation with 300 to 400 mm, which leads to annual production of 20 to 30 thousands m^3 of leachate.

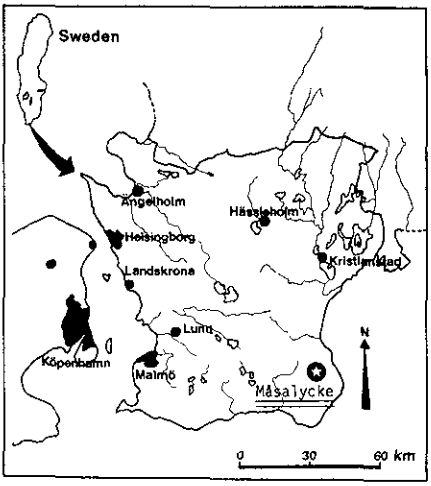

Figure 1: The location of Måsalycke in South Sweden.

In late 70's and early 80's, several accidental discharges of iron-laden leachate have been observed downstreams the landfill, and VIAK AB has been asked to suggest suitable remedial measures. Additional task was the replacement of the trucking of leachate to the nearest wastewater treatment plant by some more effective and economical, on-site treatment system. This system consists today of six steps (Figure 2):

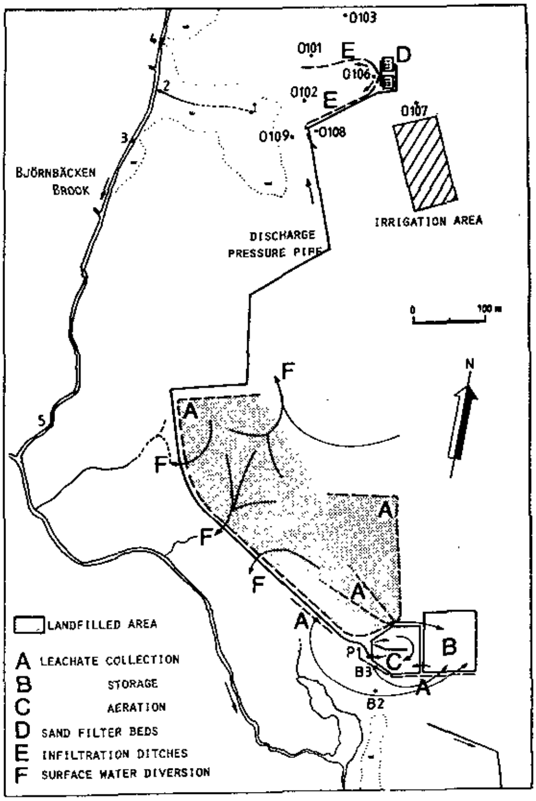

Figure 2: Måsalycke landfill (schematically).

A) Leachate collection via subsurface drains, interception ditches and pumping wells.

B) Leachate storage and flow equalization basin (volume 12.000 m^3), protecting the aeration step from sudden volume and temperature fluctuations.

C) Long-term aeration basin (volume 3.000 m^3) where leachate is oxygenated and circulated for 20 to 40 days (20 days during summer, 30 to 40 days during winter-spring). In order to optimize biological activity, phosphate is dosed manually into the basin twice a week.

D) Sandy filter beds (2 x 350 m^2), located in the forested area north of the landfill, are intended to remove iron precipitates, algae and other particles which might clog the infiltration ditches.

E) Finally, leachate is infiltrated via open ditches. During 1987, about 60 m^3/d of water was infiltrated in a 120 m long, shallow (1 m deep) open ditch. In addition, 8 to 10 m^3/d of pre-treated leachate was used for irrigation experiments at several plots vegetated with various grass and tree species and localized just south-east of sandy filter beds.

F) In order to diminish the new production of leachate, the landfill surface is successively capped with low-permeable till soil, graded and finally, when landfilling is completed, revegetated. In addition, the unpolluted surface water originating from direct precipitation or snowmelt is diverted away from the landfill by means of a superficial drainage system.

SYSTEM EFFICIENCY

Mean concentrations of some constituents in the raw leachate and consequent changes in water chemistry during the treatment are shown in Table 1. Chloride, which behaves conservatively, is more or less constant during all treatment steps with the exception of step B (dilution with direct rainfall) and E (mixing with surrounding groundwater). All other parameters (except pH and nitrate) are successively decreasing along the treatment, as a result of biochemical and physico-chemical processes such as biodegradation, assimilation, precipitation, evaporation (NH_3), nitrification (NH_4^+ to NO_3^-), adsorption, filtration etc.

Table 1: Mean chemical composition of raw leachate (A) and the leachate in consequent treatment steps (B to E, cf. Figure 2) during 1987 (A to D) and in February 1988 (E).

		A	B	C	D	E
pH		6.7	7.2	7.7	7.6	-
spec.cond. (mS/m)		114	96	88	72	14 to 18
Cl	(mg/l)	78	72	78	82	19 to 30
BOD	(")	-	45	35	7	-
COD	(")	272	165	120	94	40 to 65
NH_4-N	(")	34.5	23	3.1	2.0	0.003 to 0.03
NO_3-N	(")	0.06	0.3	11	5.7	0 to 0.5
N-tot	(")	36.5	25	18	10	0.6 to 1.2
Fe	(")	90	20	1.6	0.2	0.2 to 2
Mn	(")	16.5	6.8	2.9	0.3	0 to 0.09

It should be pointed out that, even if the rate of removal of various constituents varies between different treatment steps, none of these steps may be omitted or undersized without a serious danger of treatment efficiency losses. Parameters which should be considered when designing on-site leachate treatment systems include raw leachate quality, landfill hydrology, climatic factors, local hydrogeological conditions and the specific requirements on effluent quality with respect to possible environmental impacts.

ENVIRONMENTAL IMPACTS

The infiltrated pre-treated leachate is soon or later drained together with local groundwater flow towards Björnbäcken, a clear water brook hosting salmonid and crayfish populations, and exhibiting mean water discharge of about 100 l/s. In this brook, the direct discharge of raw, untreated leachate would theoretically raise the mean concentrations of total salts with 7.5/10%, total organics (expressed as COD) with 10/26%, total nitrogen with 29/64%, and iron with 210/490% (values for 1987 and 1986, respectively). After treatment steps B to D, the corresponding figures for 1987 are 3%, 2%, 5% and 0.3% for total salts, COD, N-tot and iron, respectively. An additional reduction of slowly degradable organics, nitrogen and other compounds occurs during step E (infiltration). Particularly important is the passage of water through peaty, water-logged areas surrounding the shores of the brook, where nitrate and some organics are removed due to denitrification processes. Other constituents such as heavy metals or chlorinated organics are very low in raw leachate at Måsalycke, and are probably removed quantitatively by the filtration, adsorption or co-metabolism processes during treatment steps C to E.

Summarized, the mean increase of concentrations in the brook is about 5% for total salts and much less for other constituents.

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WASTE DISPOSAL SITES LOCATED AT THE SEA FRONT - A GOOD ALTERNATIVE TO INLAND SITES?

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Summary

Denmark is facing increasing problems in its efforts to find new waste disposal sites. As a result, coastal sites are now being considered. A case study comprising investigation of the pollution from a coastal site is presented to illustrate possible problems resulting from such alternative locations.

The municipality of Horsens has a total population of about 50,000. The solid waste of the community has since the beginning of this century been deposited on a site which has protruded gradually into the fiord, at the head of which the site is located.

A study was undertaken with the main objective of investigating whether the marine environment of the inner fiord had suffered from the seepage from the site. The study included drilling of several boreholes on the site, a marine biological survey, and analyses of historical heavy metal concentrations of surface sediments.

The site is surrounded by an earth dam and underlain by gyttja deposits of varying thickness. The gyttja deposits are underlain by a 3-5 m thick brackish sand aquifer.

The study concluded that the pollution from the site is insignificant compared with that of the sewage treatment plant of Horsens for a majority of parameters, but conclusive evidence could not be obtained due to the general eutrophication of the fiord from the sewage discharged.

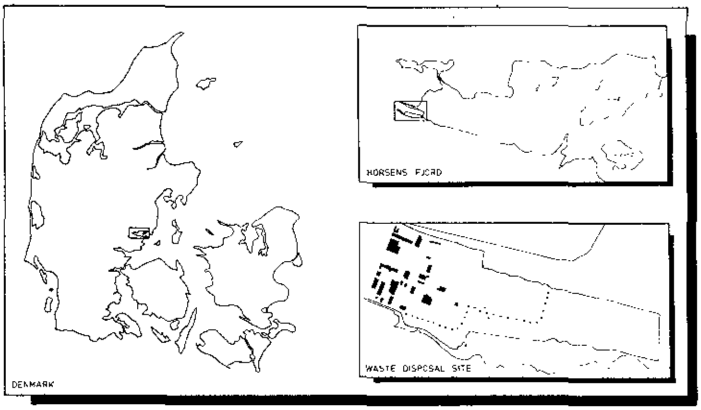

Figure 1. Location map.

1. INTRODUCTION

The town of Horsens is with its population of approx. 50,000 one of the major provincial towns of Denmark. The industrial development of the town started around 1850 and has continued since. The industrial and domestic refuse resulting from this development was until 1930 deposited at random at sites which are today for an absolute majority unknown.

From 1930, however, it appears that it has been a common practice to deposit all wastes from the town at the same location. This site is today still maintained as the public refuse site.

Horsens is located at the head of a 15 km long fiord with a relatively good water exchange. The inmost part of the fiord had shallow water depths less than 1 m before industrial development started. In the beginning of this century, however, the most shallow part of the fiord was reclaimed with sediments dredged during deepening of the harbour and a narrow navigation channel to and from the harbour.

The present public waste disposal site was commenced at the 1930 seafront. The historical site area today occupies an area of 700,000 m2 of which some 220,000 m2 is now an industrial area.

Analysis of the historical aerial photos, which are available from the area since 1945, shows the pace of refuse disposal on the site. The disposal pace was rather slow up till approx. 1965 after which a greatly accelerated pace is observed.

Disposal on the site has been uncontrolled until 1976 when a new bill imposing restrictions on disposal of hazardous waste was passed. After 1976 the site has been under supervision and only non-hazardous waste has been accepted. As a result the area deposited on up till 1976 is considered a hazardous waste site.

A municipal incineration plant was constructed on the old part of the site in 1974 and has since taken care of the flammable refuse.

Until 1975 the sewage system was discharging untreated sewage water into the fiord from numerous points. In 1975, however, a collector main and a mechanical sewage treatment plant was commissioned. The sewage treatment plant was constructed on the old part of the refuse site and consequently all sewage water has been discharged from the refuse site area since 1975.

During the seventies and eighties it became increasingly clear that the marine environment of the fiord had deteriorated to an unacceptable level. This has especially been evident in the inner fiord where a heavy eutrophication has been observed. As a result the sewage treatment plant has been extended with chemical precipitation and is now under extension with a phosphorous and nitrogen removal plant scheduled for commissioning in 1992.

It is however suspected that also the public refuse site may be contributing considerably to the pollution of the marine environment.

A study comprising a hydrogeological investigation, a marine biological investigation, and a survey of heavy metal concentrations in surface sediments was undertaken by Vejle County and Horsens Municipality to determine to what extent the site was contributing to the observed pollution of the marine environment.

2. REFUSE SITE INVESTIGATION

A total of 36 boreholes were drilled on the site. Initial research had indicated that the refuse was underlain by gyttja so some of the boreholes were drilled in pairs with one borehole only into the refuse but not penetrating the gyttja layer and another borehole close by penetrating below the gyttja layer. All boreholes were equipped with a short screen to

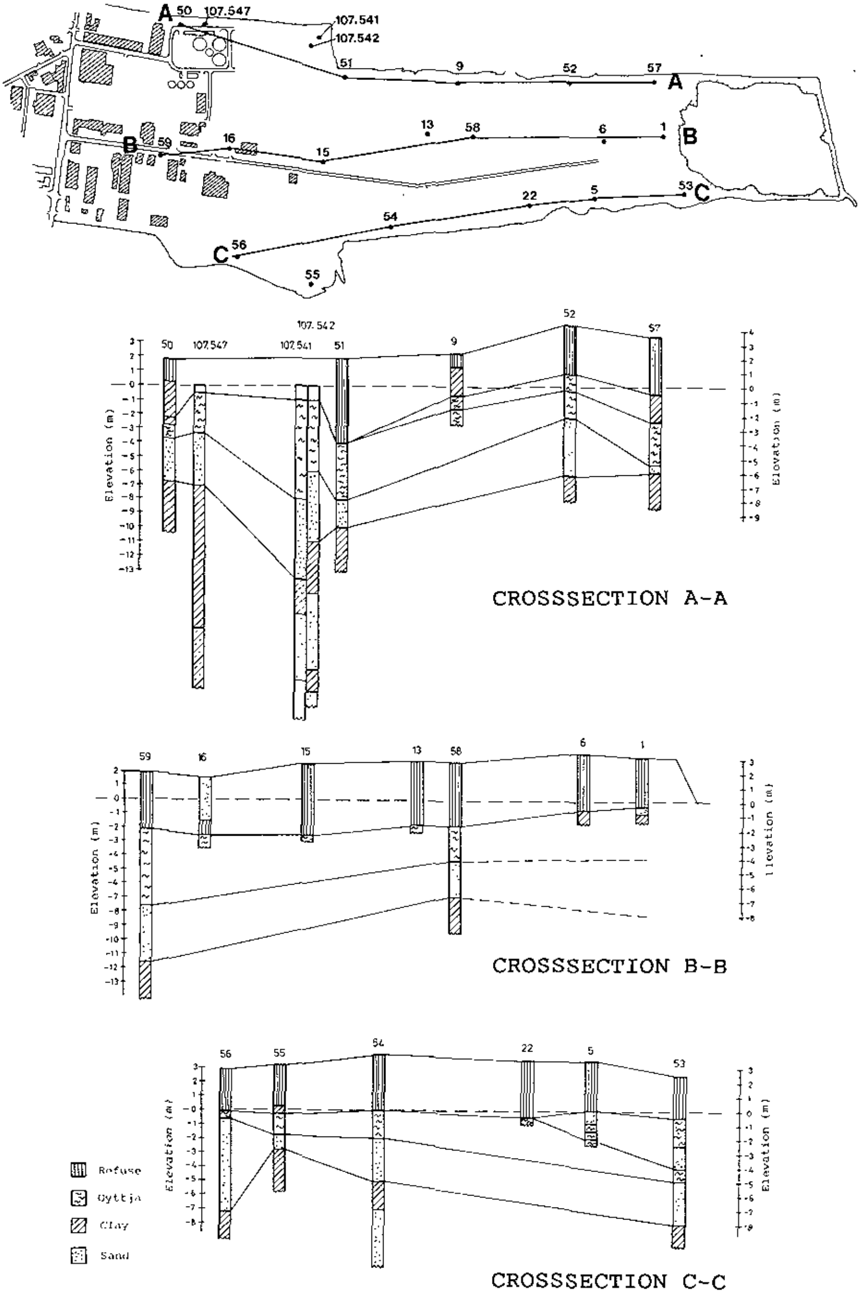

Fig. 2 Geological crosssections.

facilitate water level measurements and water sampling.

The site was found to be underlain by gyttja with a relatively high content of organic matter, with a thickness in the range of 0-7 m. The gyttja is underlain by a 3-6 m thick sand layer of preglacial origin and the sand layer is underlain by clay of same origin.

Based on analysis of historical aerial photos it was concluded that the site was surrounded by an earth dam providing at least some flow restrictions.

The ground water flow in the sand layer is north, hence confirming that no fresh water resources are in jeopardy. The groundwater elevation is 0.3-0.6 m.

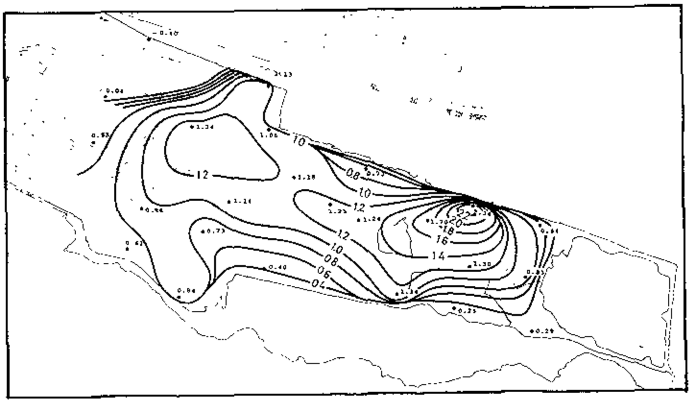

Fig. 3. Leachate elevations

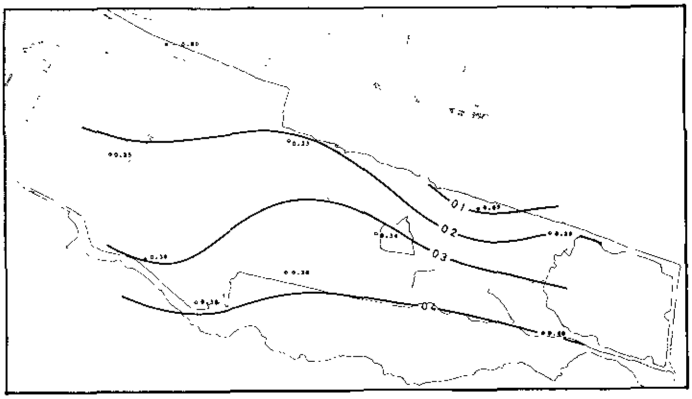

Fig. 4. Water levels in sandlayer below gyttja deposits.

The leachate water level is with one local exception found at elevations 0.3-2.4 m. For the greater part of the site this means that the leachate elevation is approx. 1 m higher than the water level elevation in the sand layer below the gyttja. Hence it could be concluded that flow from the site is both vertical through the gyttja and horizontal through the earth dam.

Leachate elevations along the southern perimeter are more or less identical to the elevations in the sand layer hence indicating that vertical flow is more or less unrestricted in this area. This was found to coincide with the area with no gyttja below the refuse and hence indicates flow below the dam in that area. Elevations below sea level were recorded in the north-western part of the site but this was caused by a temporary lowering of the water level due to construction work at the sewage treatment plant.

Water samples were collected from 20 boreholes and analysed for heavy metals, phosphorus, nitrogen, and TOC. The analysis results appear from Table 1.

	Leachate			Sandlayer			Sewage	
	Total	Hazard. site	Non-hazard. site	Total	Hazard. site	Non-hazard. site	Present	Future max. permissible
No of boreholes	11	6	5	9	4	5		
Mercury		<0.0002			<0.0002		0.35	-
Chromium	0.063	0.038	0.097	0.028	0.027	0.030	<0.1	0.2
Zinc	0.770	0.360	1.348	0.290	0.200	0.356	1	1
Copper	0.450	0.280	0.694	0.087	0.090	0.088	0.37	0.2
Nickel	0.179	0.140	0.236	0.080	0.074	0.085	<0.1	0.2
Lead	0.320	0.230	0.438	0.170	0.117	0.166	<0.1	0.2
Cadmium	0.047	0.046	0.048	0.025	0.023	0.028	<0.1	-
Arsenic	0.054	0.040	0.073	0.005	0.003	0.007	-	-
Phosphorus	2.9	1.3	5.0	0.25	0.27	0.22	6.7	1.0
Nitrogen	97	134	4.6	2.5	3.2	0.4	49	6
COD	44	457	42	58	32	79	600	-
TOC	103	72	141	1.0	1.3	0.9	-	-
Phenol	0.021	0.029	0.019	0.022	0.033	0.013	-	-

Table 1. Average solute concentrations in leachate and sandlayer, results of one analysis of sewage inflow to sewage treatment plant and future discharge criteria.

Solute concentrations were found to be of the same magnitude as concentrations in the sewage water inflow to the sewage treatment plant and concentrations were found to be highest in the youngest refuse. Some heavy metal concentrations are lower in the sand layer due to the dilution effect and due to retention in the gyttja. The dilution effect appears to result in a halving of concentrations as this is observed for cadmium, which is known to be unaffected by a colloidal bonding. The retention effect is most significant for arsenic and copper. Cadmium, lead, nickel and chromium do not appear to be retained.

Phosphorus and nitrogen concentrations in the sand layer is 9% and 3% respectively of the leachate concentrations and hence of the same magnitude as recorded previously in the waters of the head of the fiord.

The average outflow from the refuse site area was based on water budget considerations computed to max. 475 m3/day or 2.4% of the discharge from the sewage treatment plant. The groundwater flow below the site was estimated to be within the range of 65-650 m3/day.

3. MARINE BIOLOGY

The marine-biological investigation included preparation of a solute budget for the fiord from various existing sources. (Table 2 and 3). The budget indicates that the refuse site is responsible for 17% of the cadmium load of the fiord. Other main contributors are chromium with 9% and nickel with 10%. All three figures are of the same magnitude as the atmospheric contribution.

The total phosphorus and nitrogen contribution to the fiord is less than 1%.

Bioaccumulation measurements included measurements of lead and mercury in mussels on a location at the refuse site and at a reference station in the outer part of the fiord. For lead no significant differences were detected, but the measurements indicated that lead concentrations are 10 times higher than previously measured in the Little Belt in the inner Danish waters.

Mercury concentrations at the refuse site were found to be twice as high as the concentrations measured at the reference station.

	Contribution in %					
	Cadmium	Chromium	Copper	Nickel	Lead	Zinc
Waste disposal site	17	9	3	10	3	1
Bygholm stream	25	38	4	37	3	10
Hansted stream	23	35	3	34	3	9
Remaining catchment	-	-	-	-	-	-
Sewage treatm.plant	19	8	84	3	68	68
Other treatm.plants	1	0	3	0	2	2
Atm. precipit.	15	10	3	16	21	9
Miscellaneous	-	-	-	-	-	-
Total percentage	100	100	100	100	100	99
Total kilos	48	118	2581	312	882	9085

Table 2. Heavy metal budget for Horsens fiord. (Ref. 1).

	Contribution in %		
	Nitrogen	Phosphorus	COD
Waste disposal site	1	0,3	1
Bygholm stream	29	19	15
Hansted stream	20	11	13
Remaining catchment	22	5	13
Sewage treatm.plant	21	57	53
Other treatm.plants	1	5	5
Atm. precipit.	5	1	-
Miscellaneous	1	2	-
Total percentage	100	~100	100
Total kilos	1.650.000	130.000	6.350.000

Table 3. Nutrient budget for Horsens fiord. (Ref. 1)

Acute toxicity tests were performed on the bottom crustacean Nitocra Spinipes with leachate chate from 5 shallow boreholes. The tests indicated mortal effects only for concentrations stronger than approx. 1:4.

A vegetation survey of the fiord was performed. The survey indicated a water visibility of only 0.7 m near the refuse site which gradually increased to a maximum of 3.6 m approx. 6 km from the site. It was concluded that the poor visibility near the site was caused by the heavy eutrophication of the fiord.

A survey of bottom fauna was performed. The survey indicated dominance of pollution resistant individuals near the refuse site. Comparison of fauna compositions in 4 stations along the southern perimeter of the site indicates seepage of fresh water with a high organic content along the eastern part of the southern perimeter. This was found to be in agreement with the findings of the hydrogeological survey.

4. SEDIMENT ANALYSES

Analyses of historical fluctuations in heavy metal concentrations were performed by means of Pb-210 measurements on a 28 cm long column taken from the marsh areas south of the refuse site, Table 4.

However, due to a very low accumulation rate this method can only be used for sediment dating within the last 100 years. The results of

	Period				
Metal	1898-1923	1838-50	1961-68	1974-79	1983-87
Mercury	1.3		1.2		1.0
Lead	1.8		1.6		1.5
Cadmium	7.6		2.4		0.8
Zinc	11.3	7.4	6.0	3.9	3.2
Chromium	1.1	0.9	0.9	0.9	0.8
Nickel	4.8		2.7		1.5
Arsenic	8.2	4.9	4.3	3.0	2.5

Table 4 Heavy metal enrichment factors determined from analyses of concentrations in one column. (Ref. 2).

the dating and analyses for heavy metal concentrations appear from table 4 as enrichment factors compared to the levels more than 100 years ago.

It appears that all heavy metal concentrations increased until the beginning of this century and have then been declining gradually since. Fluctuations in concentrations of zinc, cadmium, arsenic, and nickel are most significant.

Several factors may have contributed to this. One likely explanation is the fact that harbour dredging started around 1870. Most of the dredged sediments were deposited near the present refuse site. However, what is important for the investigation, is the fact that no increase in heavy metal concentrations has been detected during the lifetime of the refuse site.

5. REMEDIAL ACTIONS

Two alternative remedial action systems were outlined for further discussion.

The first system comprises a number of wells dug in the refuse to the top of the gyttja, Figure 5. *By controlled pumping from these* wells with pumping water levels around fiord sea level, it will be possible to greatly reduce leachate seepage to the fiord. The percolate will then be treated in the existing sewage treatment plant or in a separate minor plant.

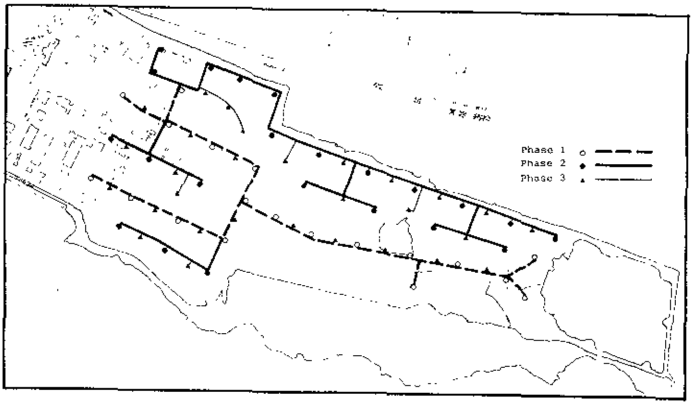

Figure 5 Remedial action system, alternative 1.

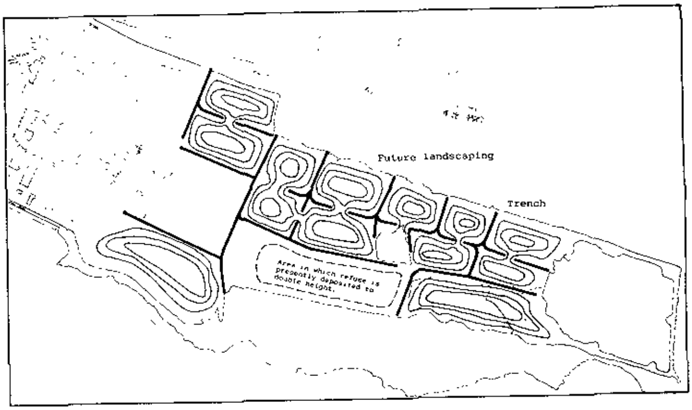

Figure 6 Remedial action system, alternative 2.

An alternative system comprises reduction of percolate production by landscaping and vegetation to facilitate surface runoff and evaporation. This alternative further opens for the possibility of further depositing to a higher elevation.

6. CONCLUSION

The coincidence of sewage discharge area and leachate seepage made it impossible to conclude from several of the marine-biological elements of the investigation that the refuse site presents an unacceptable load to the fiord.

A comprehensive water environment plan will be implemented in Denmark within the next 5 years and will result in reduction of nutrient loads from sewage and agricultural activities. This will of course result in a larger percentage of load from the leachate. The magnitude of this percentage is presently being analysed.

However, it remains a fact that the seafront location of the waste disposal site has been fortunate in respect of protection of fresh water resources.

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IMPACTS OF SOLID AND LIQUID WASTES ON GROUND AND SURFACE WATER: A HYDROCHEMICAL APPROACH

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Summary

Hydrochemical study is regularly carried out on both the ground and surface water samples from various disposal sites in the Madras City and its environs since 1985. Three main waste disposal sites are considered and discussed here for their impact on surface and ground water chemical quality variations:

1. Solid waste disposal site near the Foreshore Estate, Mandaveli, Madras.
2. Liquid waste disposal through the Cooum River near the Island ground, Madras.
3. Brine and estuarine water zone near the Muttukkadu estuary, Chingleput District, Tamil Nadu.

pH, temperature, electrical conductivity, and major ion concentrations(Na^+, K^+, Ca^{++}, Mg^{++}, Cl^-, HCO_3^-, NO_3^-, $F^-$and $SO_4^=$) were determined in each sample and water level fluctuations in wells were also monitored.

In all these three sites, positive correlation between the surface and groundwater is recognised and also a clear cut seasonal variation is seen i.e., all the major ion concentration show an increase upto the beginning of the monsoon and then a decrease. Though the surface water is not suitable even for bathing in these areas, the groundwater adjoining these areas are somewhat good for drinking purposes except the well located around the solid waste disposal site. The main factors for the above mentioned contrast might be the dilution and filtering effects in the case of liquid wastes whereas the concentration and leaching of pollutents in the case of solid wastes.

In conclusion either it is a solid waste or a liquid waste, it should be properly treated and disposed to save our environment and future generation.

INTRODUCTION

All over the World the monsoon fails frequently and in many place for subsequent years. Consequently availability of surface water becomes scarce or even nil. Hence many cities in the World utilise groundwater for domestic water supply, industries and agricultural practices because it is a replenishable resource and also it has got inherent advantages over surface water. But the inherent advantages of groundwater are often changed or even nullified by the mismanagement of solid and liquid wastes near these water reservoirs by the private parties and as well as Government agencies. In this article, the impacts of solid and liquid wastes on ground and surface waters are explained with a few examples from(in and around) Madras city, India.

STUDY AREA

Hydrochemical studies were carried out in and around the South Madras Coastal Aquifer (SMCA) which is enclosed by the Cooum River in the North, Muttukadu estuary in the South, Bay of Bengal in the East and Buckingham Canal in the West. The aquifer of this area along with adjacent locations(north and north west) is commonly known as the MADRAS AQUIFER SYSTEM - and this system is regularly monitored for its hydrogeochemistry since 1985.(Elango and Manickam, 1986; 1987 a and b, Elango, Manickam and Sakthivadivel, 1988). The Cooum River is almost a dead river and this serves atmost as the city's sewer drain. The Ady River flowing across the SMCA is seasonal and having flow only a few days during the northeast monsoon period of October to December. The Muttukadu estuary cut across the Buckingham Canal which runs parallel to the coast, once served as inland navigation canal, now transports only the waste and sewers. The average annual rainfall over this area is about 1230 mm(Bangarraju, 1988). This semi-confined shallow aquifer(240 km^2) exists in the Quaternary coastal sandy sediments with thickness ranging from 7 to 12 m and is underlain by impermeable charnockite basement rock.

Three main waste disposal sites are considered and discussed here for their role on the chemical quality variations of ground and surface waters(Fig.1).

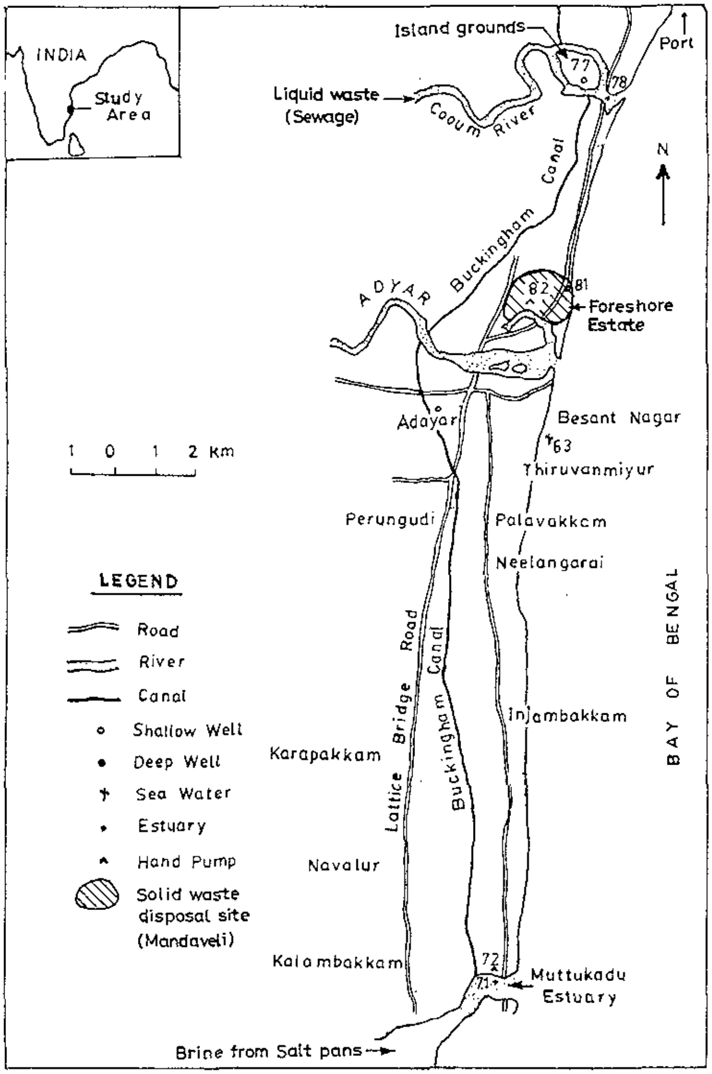

Fig.1 Sampling locations in and around South Madras Coastal Aquifer, India

1. Solid waste disposal site near the Foreshore Estate, Mandaveli, Madras.
2. Liquid waste disposal through the Cooum estuary near the Island ground, Madras.
3. Brine and estuarine water zone, Muttukadu estuary, Chingleput District, Tamil Nadu.

METHODOLOGY

Both surface and sub-surface water samples were collected on a monthly basis between July 1987 and January 1988 from the sampling sites mentioned in the previous paragraph. Surface samples were taken from the estuaries and from the sea and the sub-surface samples were from shallow wells($\angle$ 10 m deep) and hand pumps. The temperature(air and water), pH, electrical conductivity(Ec) of the water samples were found out in situ. The water level of the wells and the general conditions of the disposal sites were also noted at every trip. The samples were analysed in the Hydrochemistry laboratory of the Centre for Water Resources, Anna University, Madras for the concentrations of major ions, Na^+, K^+, Ca^{++}, Mg^{++}, Cl^-, HCO_3^-, CO_3^{--}, SO_4^{--}, NO_3^- F^-. The ion concentrations of Na^+, K^+, Cl^-, NO_3^- and F^- were measured using ORION Specific Ionalyzer - 901 and Ion Selective Electrodes. On certain ocasions, Na^+ and K^+ values were also determined by ELICO flame photometer. The concentrations of Ca^{++}, Mg^{++} and carbonates were found out by volumetric titrations. The SO_4^{--} content alone was determined with a SYSTRONICS spectrometer(Model 106). The theory and principles of the chemical analyses are detailed elsewhere(Standard Methods, 1971; Suess, 1982).

RESULTS AND DISCUSSION

The water level in wells was very low during the month of September and it raised to a maximum in January. The steep rise from the month of October was due to recharge of rain water brought by the north-east monsoon and reduced uptake of groundwater (BangarRaju, 1988).

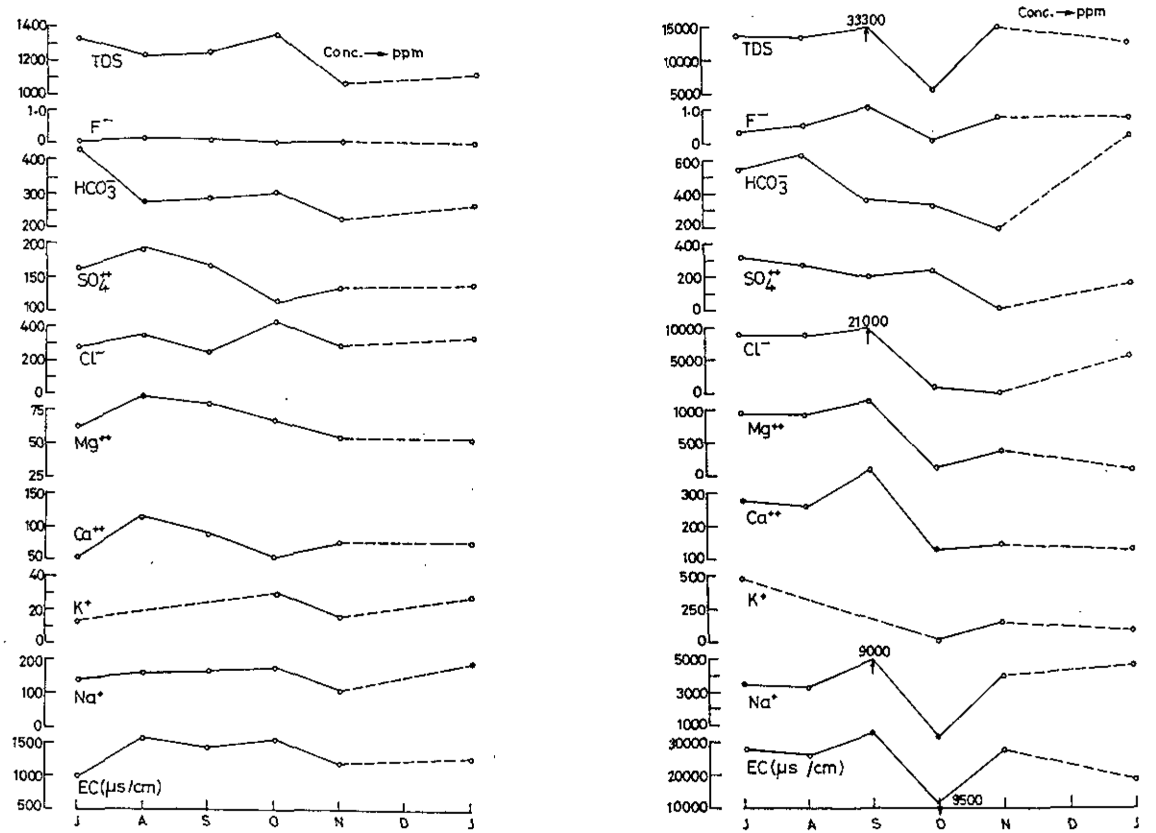

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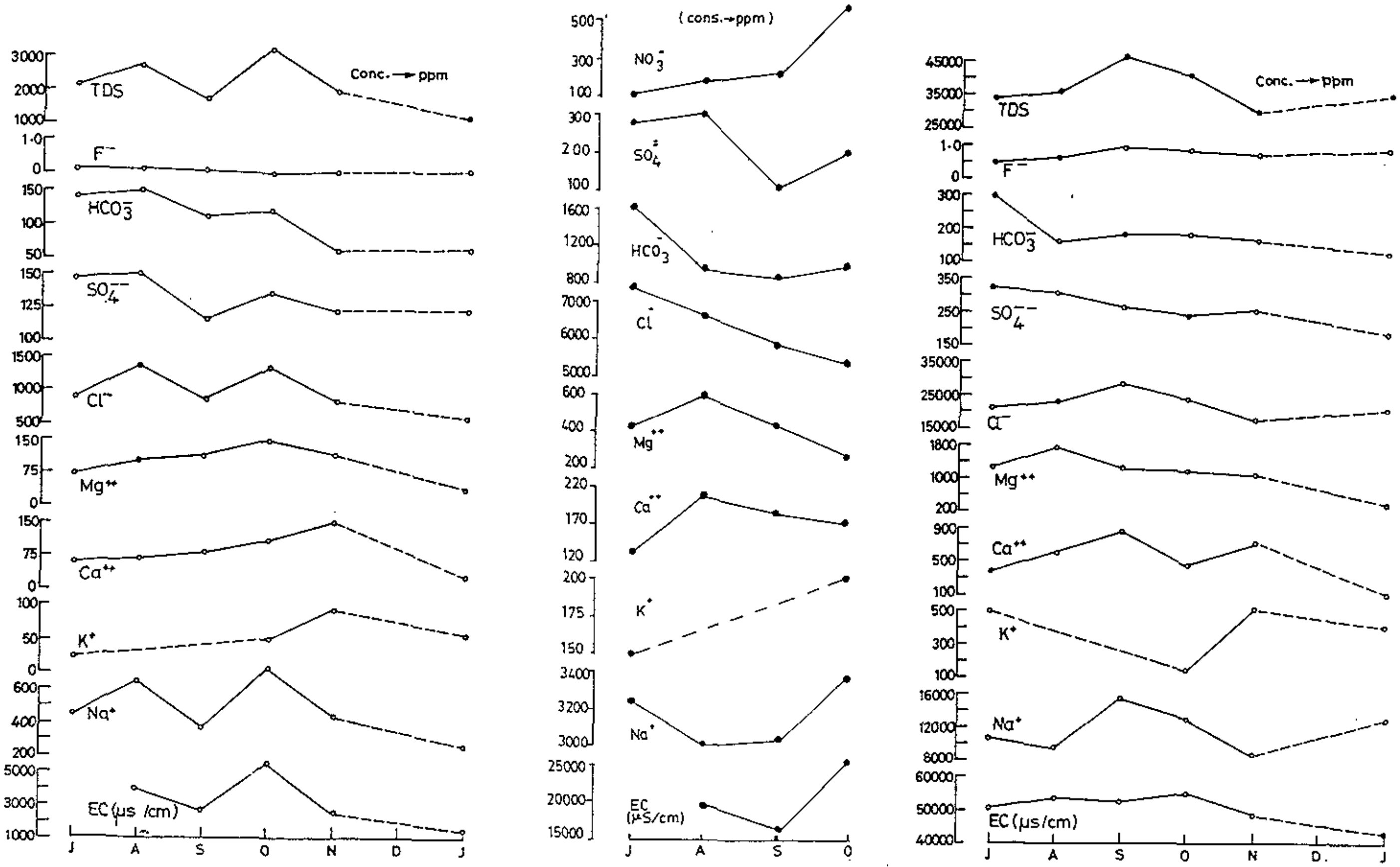
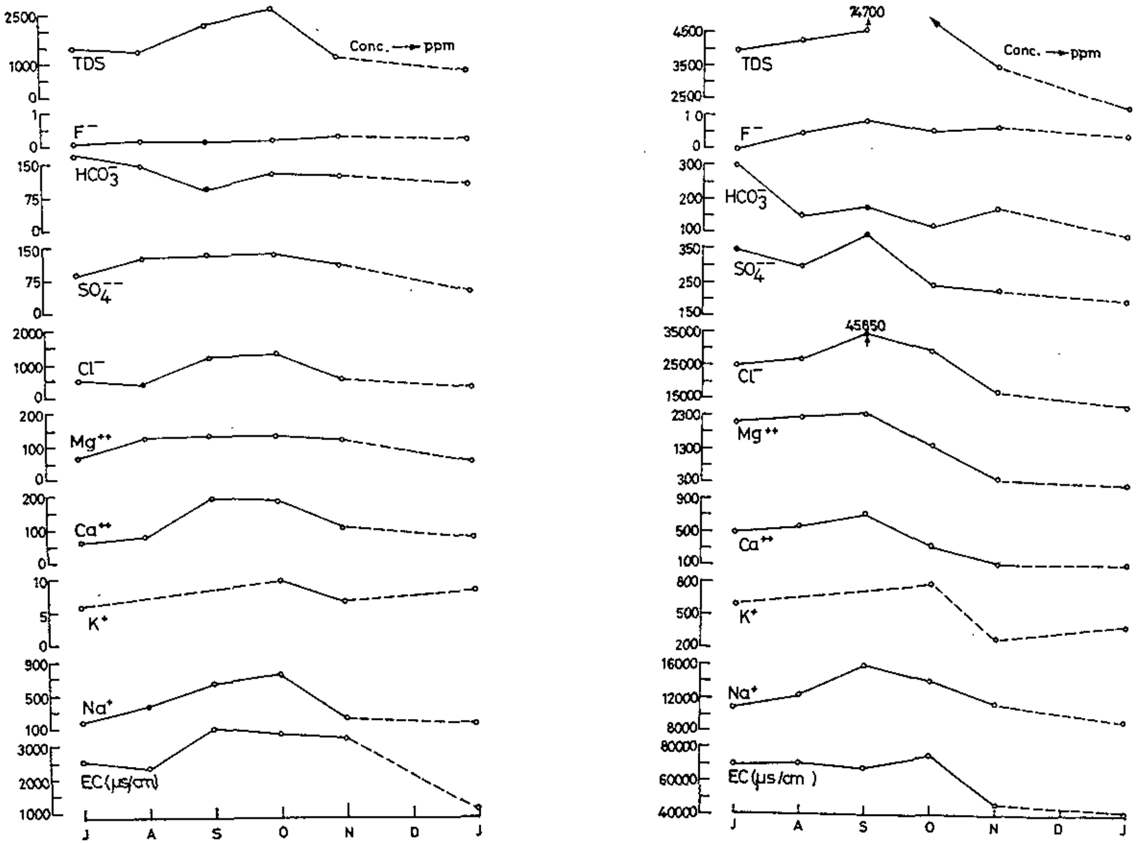
Fig.2 Temporal variation in hydrochemical quality(1987-1988). a, Dug well, Island Ground, Madras(S.No.77). b. Cooum estuary - sewage(S.No.78)



3a 3b 3c

Fig.3 Temporal variation in hydrochemical quality(1987-1988). a, Dug well with electric pump, Foreshore estate(S.No.81). b, Hand pump near solid waste disposal site, Mandaveli (S.No.82). c, Sea water(S.No.63)



4a 4b

Fig.4 Temporal variation in hydrochemical quality(1987-1988). a, Hand pump on the bank of Muttukadu estuary(S.No.72). b, Muttukadu estuary-brine mix from salt pans(S.No.71)

Seasonal variation in hydrochemistry

The temporal fluctuations in major ion concentrations, between July 1987 and January 1988 for ground and surface waters are shown in Figs.2-4. The hydrochemistry of surface and sub-surface waters vary significantly with time. These variations in major ion values are mostly in accordance with the variation in electrical conductivity and they show an increase in their value upto the beginning of the monsoon and then a decreasing trend due to dilution.

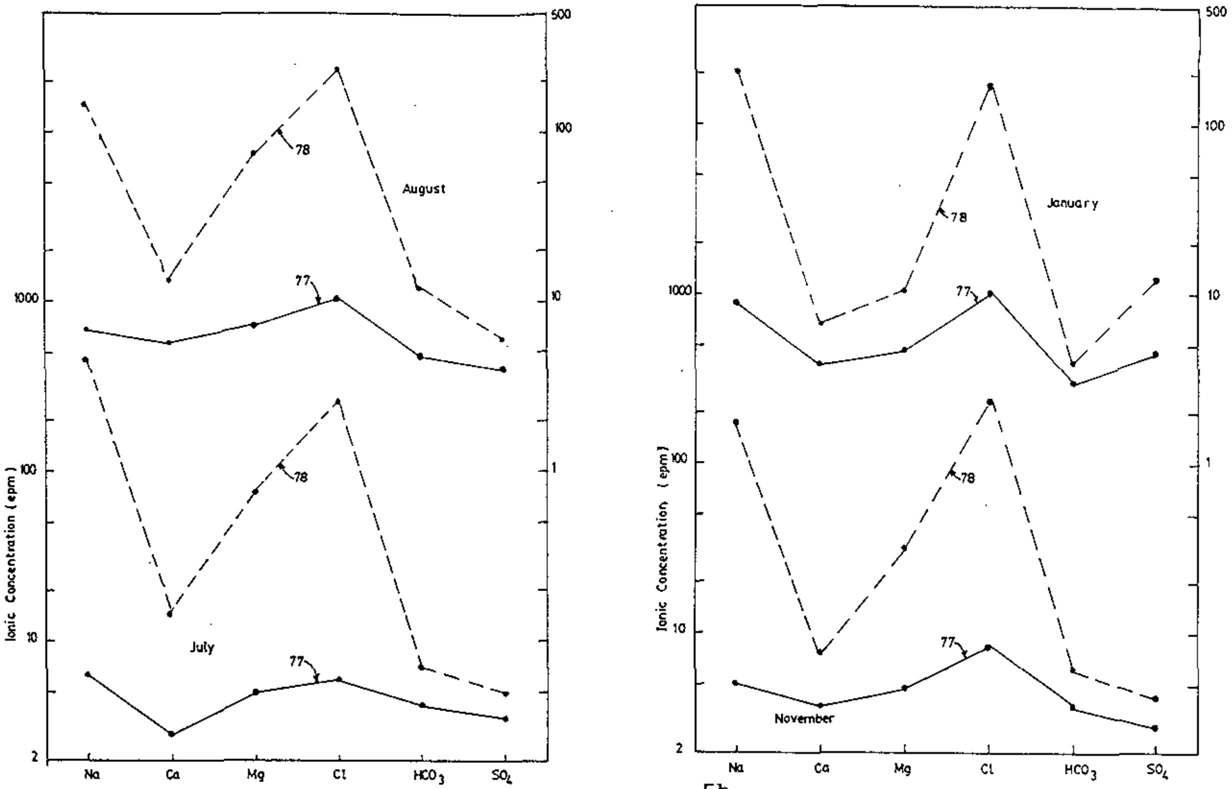
Comparison between ground and surface water chemistry

Schoeller semi-logarithimic diagrams are used to correlate the following three sets (Table 1) of ground and surface water samples (Figs.5-7).

Table 1. ***Surface and subsurface water sample locations and the major source of contamination***

S. No.	Ground-water/well type	S. No.	Surface water	Source of contamination
77	Island Grounds/dug well	78	Cooum Estuary	Liquid waste (sewer)
81	Foreshore Estate/dug well	63	Sea	Solid waste (carbage)
82	Mandaveli/ hand pump		"	"
72	Muttukadu/ hand pump	71	Estuarine & brine waters	Brine from nearby salt pans

Here the continuous line represents the groundwater chemical quality and the dashed line represents the surface water chemistry. The ratio of the ions in both analyses are equal since lines joining any two ions are parallel and as they were plotted to logarithmic scale. Hence, there is good correlation among the above mentioned three sets of ground and surface waters. Thus the contamination of groundwater by the surface water and in turn by the solid and liquid wastes is confirmed.



5a

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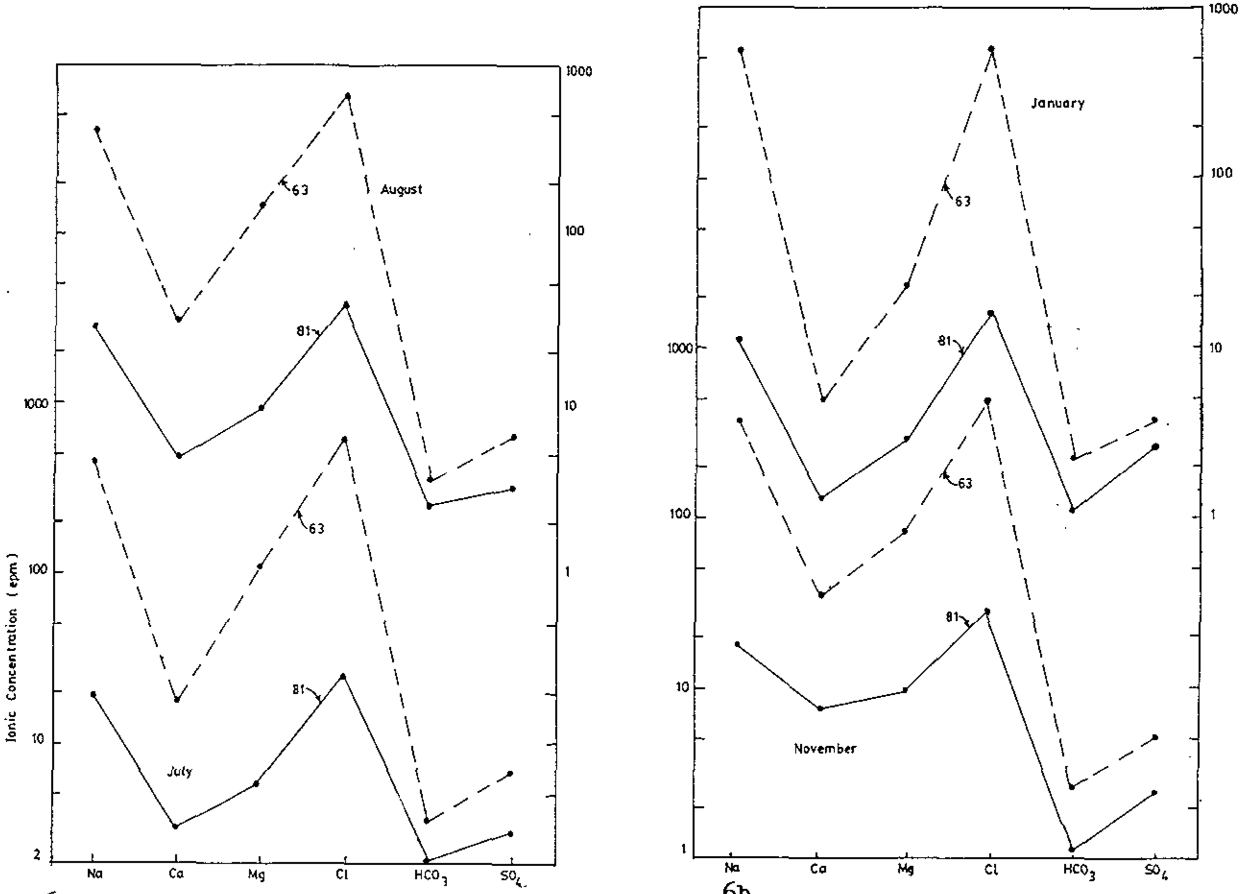
Fig.5 Schoeller's correlation graph between S.Nos.77 and 78. a, Before monsoon-July and August,1987. b, After monsoon-November, 1987 and January, 1988



6a 6b

Fig.6 Schoeller's correlation graph between S. Nos.63 and 81. a, Before monsoon-July and August, 1987. b, After monsoon-November, 1987 and January, 1988

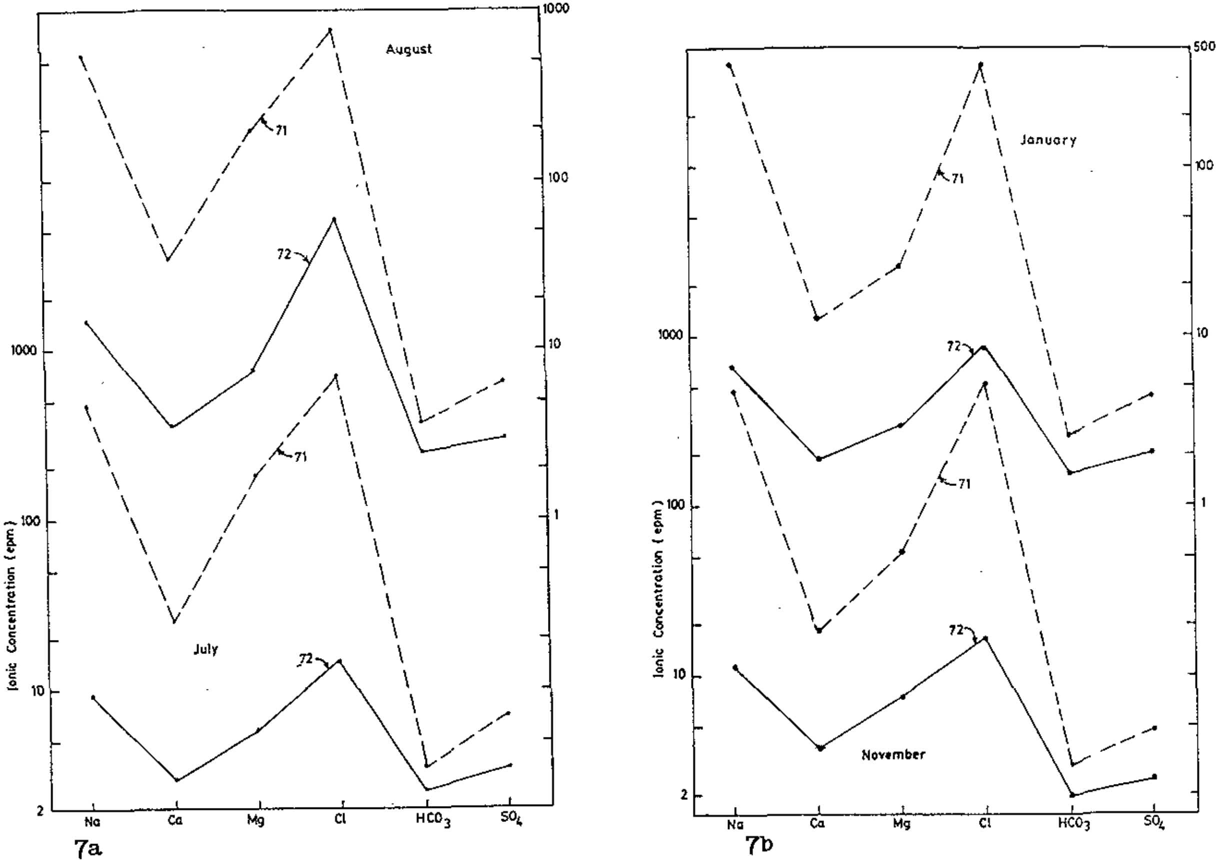


Fig.7 Schoeller's correlation graph between S.Nos.71 and 72. a, Before monsoon-July and August, 1987. b, After monsoon-November, 1987 and January, 1988

Water quality considerations

The surface water is not suitable either for bathing or for irrigation purposes in these areas(Table 2) whereas the groundwater adjoining these areas is somewhat good for drinking purpose except the well located around the solid waste disposal site(Table 3). This is more clear when we consider the seasonal variation in hydrochemistry of the wells(Figs.2-4). The well waters situated near the liquid wastes are not potable only during the severe summer whereas the well water located near the solid wastes is not suitable throughout the year(Table 3).

Table 2. SAR classification and irrigation water tolerance limits

SAR classification	SAR value	Site numbers Ground water	Site numbers Surface water
Excellent	<10	72,77	-
Good	10-18	81	-
Fair	18-26	-	-
Poor	>26	82	63,71,78

Parameter	Tolerance limits	Site numbers* Ground-water	Site numbers* Surface water
Ec(m.mhos/cm)	3.0	72,81,82	63,71,78
Cl^-(mg/l)	600.0	72,81,82	63,71,78
SO_4^{--}(mg/l)	1000.0	-	-
Na^+(%)	60.0	81,82	63,71,78

* in which limit exceeds

The main factors involved in affecting the groundwater by the liquid and solid wastes are:

1. Liquid wastes get diluted during the rainy season by fresh surface water and the suspended particles settle down by differential settling phenomenon(Manickam, Barbaroux and Ottmann, 1985) and making a impermeable layer in the bottom which in turn prevents or slow down the process of infilteration into the groundwater.
2. On the contrary, the solid wastes do not get compacted very easily and the rain water or any other water body in contact

Table 3. Comparison with drinking water standards

Parameter	Standards			Groundwater		
	India*	Europe**	U.S.A.**	Min.	Max.	S.No.***
pH	6.5-9.2	6.5-8	6.5-8	6.91	7.90	-
Ec (m.mhos/cm)	-	-	-	0.80	25.50	-
Na^+(mg/l)	-	175	200	106	3350	72 - 81,82
K^+(mg/l)	-	12	-	6.0	200	- 77,81,82
Ca^{++}(mg/l)	200	-	200	26.6	212	- - - 82
Mg^{++}(mg/l)	150	125	125	11.4	592	- - 81 82
Cl^-(mg/l)	1000	200	250	264	7472	72 - 81 82
HCO_3^-(mg/l)	-	-	500	64	1598	- - - 82
SO_4^{--}(mg/l)	400	250	250	50	300	-
F^-(mg/l)	1-2	0.6-1	0.6-1	0.05	0.50	L L L L
Hardness(mg/l)	600	500	-	209	2965	72,77,81,82
Alkalinity (mg/l)	-	-	-	52	965	-

* taken from the Manual(1963)
** taken from Mathess(1982)
*** in which limit exceeds L - lesser than the recommended limit.

simply dissolves the pollutents and takes them to the subsurface water.

Therefore the impacts of solid wastes on ground or surface water is much more than the liquid wastes. It does not mean one can simply dump the liquid wastes into the surface or subsurface water bodies. All the wastes should be properly treated and disposed to save our environment and our future generation.

CONCLUSIONS

From the hydrochemical study carried out in and around the South Madras Coastal Aquifer by systematic collection and analyses of surface and subsurface water samples from the waste disposal sites, the following conclusions have been arrived at:

1. The chemical quality of both surface and groundwaters is much influenced by the north-east monsoon i.e., all the major ion concentrations increase until the onset of the monsoon and then show a decreasing trend.
2. A positive correlation exists between the ground and contaminated surface waters by the wastes. Hence the impacts of wastes on surface and sub-surface waters is established.
3. The solid waste has more effects on water bodies than the liquid wastes. The main factors for this contrast might be the dilution and settling phenomena in the case of liquid wastes whereas the concentration and leaching of pollutents in the case of solid wastes.

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INDIRECT MAPPING OF SOIL POLLUTION BY DETECTION OF VOLATILE ORGANIC COMPOUNDS

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INTRODUCTION

This paper presents a case of soil pollution involving volatile organic compounds used in chemical laundry and in cleaning industries.

Pollution in the ground is often observed in connection with excavations for new buildings. This typically calls for a risk analysis, a sufficient mapping of the pollution and for averting measures. The key words in these measures are normally economy and speed, especially when ground water resources are concerned, or when the future use of the site, e.g. residential quarters, is very sensitive to pollution. Another key word is confidence in the chosen measures. One thing is, what the responsible experts can agree upon with the authorities, another what the public opinion will be.

In this case we had an option of stripping the soil for its pollution in situ, but even though a considerable amount of polluted soil was chosen to be removed. The former industry found this the fastest and alo the safest way of keeping its good image. The in situ treatment was found premature with respect to what the public opinion would be.

The measures, with emphasis on risk analysis and the pollution mapping, are presented below by the responsible N&R Consult, and the joint venturer Technological Institute. The latter performed the measurements of soil air and soil surface emmisions.

CASE STORY

The site of 2500 m^2 in the centre of the city Aalborg, Denmark, has for the major part of this century been used by laundry and cleaning works. The former buildings were removed in 1986 to give place to apartment buildings. Under the excavations for these buildings soil and ground water pollutions were observed. The substances were those used by the former industry, mainly tetrachloroethene (TeCE) and trichloroethene (TriCE) and some turpentine (Tu).

A first investigation was then carried out mainly in one third of the area, where hot spots were expected according to the location of the industrial installations. It was a typical hydrogeological approach with soil and water sampling from 18 shallow and 3 deep boreholes.

At this stage the authors were engaged in the case to determine the pollution boundaries more precisely, and to asses the environmental impact by the future use of the area. At the end of 1987 the whole area was mapped indirectly by measuring soil air concentrations and emissions of the involved volatile organic compounds. At the same time the critical concentrations in the ground water and the atmosphere were determined through a toxicological study.

Based on the investigations the averting measures for the ground water pollution could immediately be decided to be a pumping from some abstraction wells at the hot spot area for about 3 months. The water is stripped in a column, and the venting air cleaned through a activated carbon filter. The ground water clean up began last week.

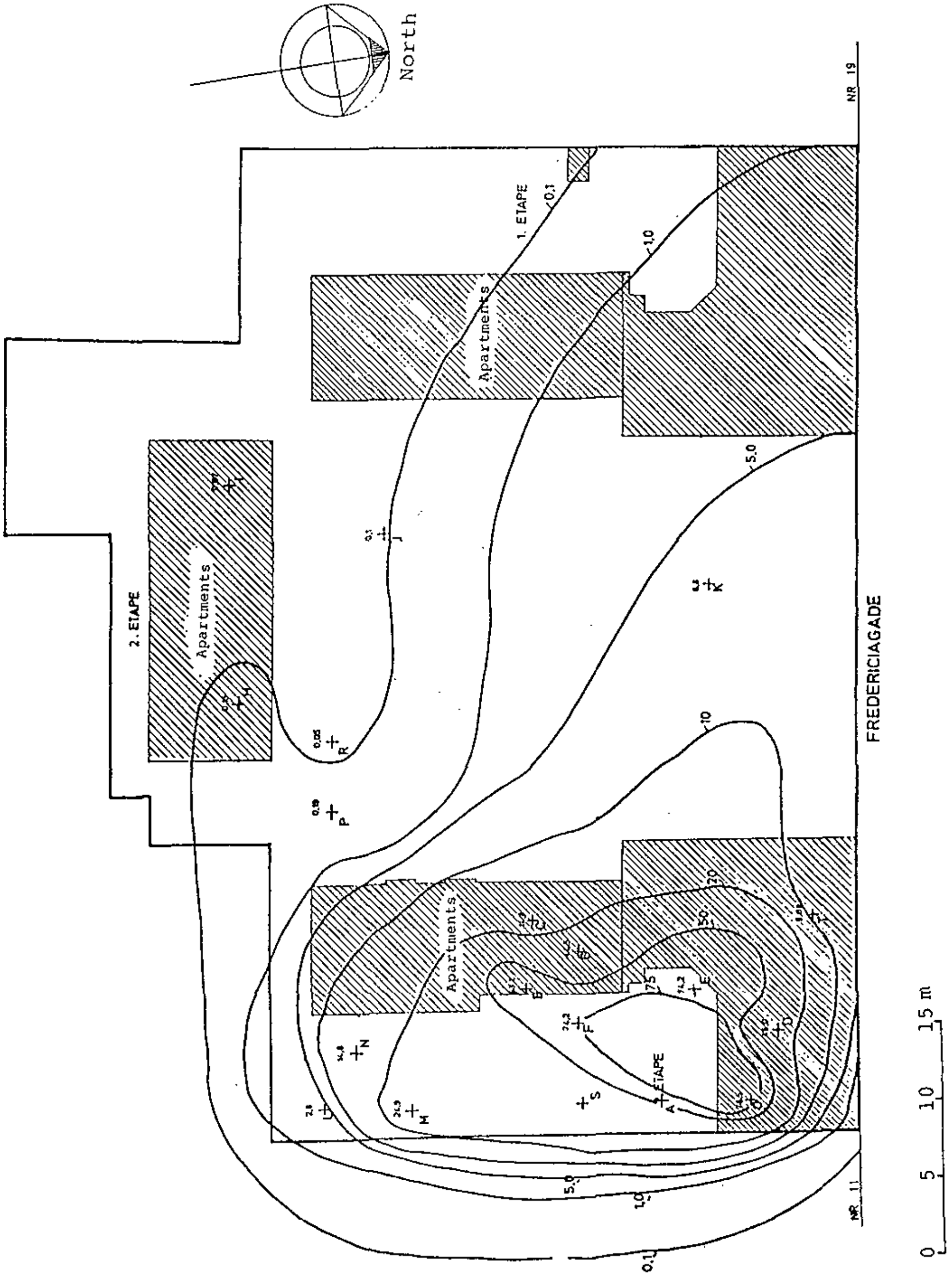
Based on the assessment of emission and the toxicological evaluation a removal of contaminated soils in about 20% of the area down to the ground water level in 2 m's depth was also carried out.

IMPACT ON THE GROUND WATER RESERVOIR

The impact on the ground water was assessed through the available hydrogeological informations from this part of town as well as the detailed local results from the boreholes. Under the 1-2 m's overburden of sandy mould and fill are up to 40 m's layers of fine sands with minor bodies of gravel or clay/silt. Beneath the sands is chalk partly protected by a thin layer of clays. This gives only one ground water body in a phreatic aquifier consisting of sands above and chalk below, which is subject to abstraction for both industrial use and drinking water. The ground water level is at about two meters depth and the flow is to the north to af fjord about 1 km away.

The pollution was measured in the aquifer to be mainly at the hot spots where installations have been placed. The penetration depth observed was 15 m with soil sample concentrations of 415,7 mg TeCE/kg and 0,37 mg TriCE/kg. The concentrations were varying much with depth due to variations in clay contents. Normally the penetration depth for TriCE was found less than for TeCE, being of the order of 5 m. Maximum concentrations were

Figure 1. TeCE-content in ground water (mg/l)



generally observed at about 4 m depth with up to 1800 mg TeCE/kg and 125 mg TriCE/kg, the latter being exceptional for TriCE. Typical high levels were of the order of 5-50 mg TeCE/kg and 0,5-5 mg TriCE/kg.

Water samples from the 21 boreholes as well as 10 industrial boreholes down streams, showed high levels in 30% of the boreholes sampled, being 40-80 mg TeCE/l on the site and 15-40 µg TeCE/l down streams. The solubility (saturation) is not reached in any place. The results are shown in figure 1.

IMPACT ON THE SOIL AND ATMOSPHERE

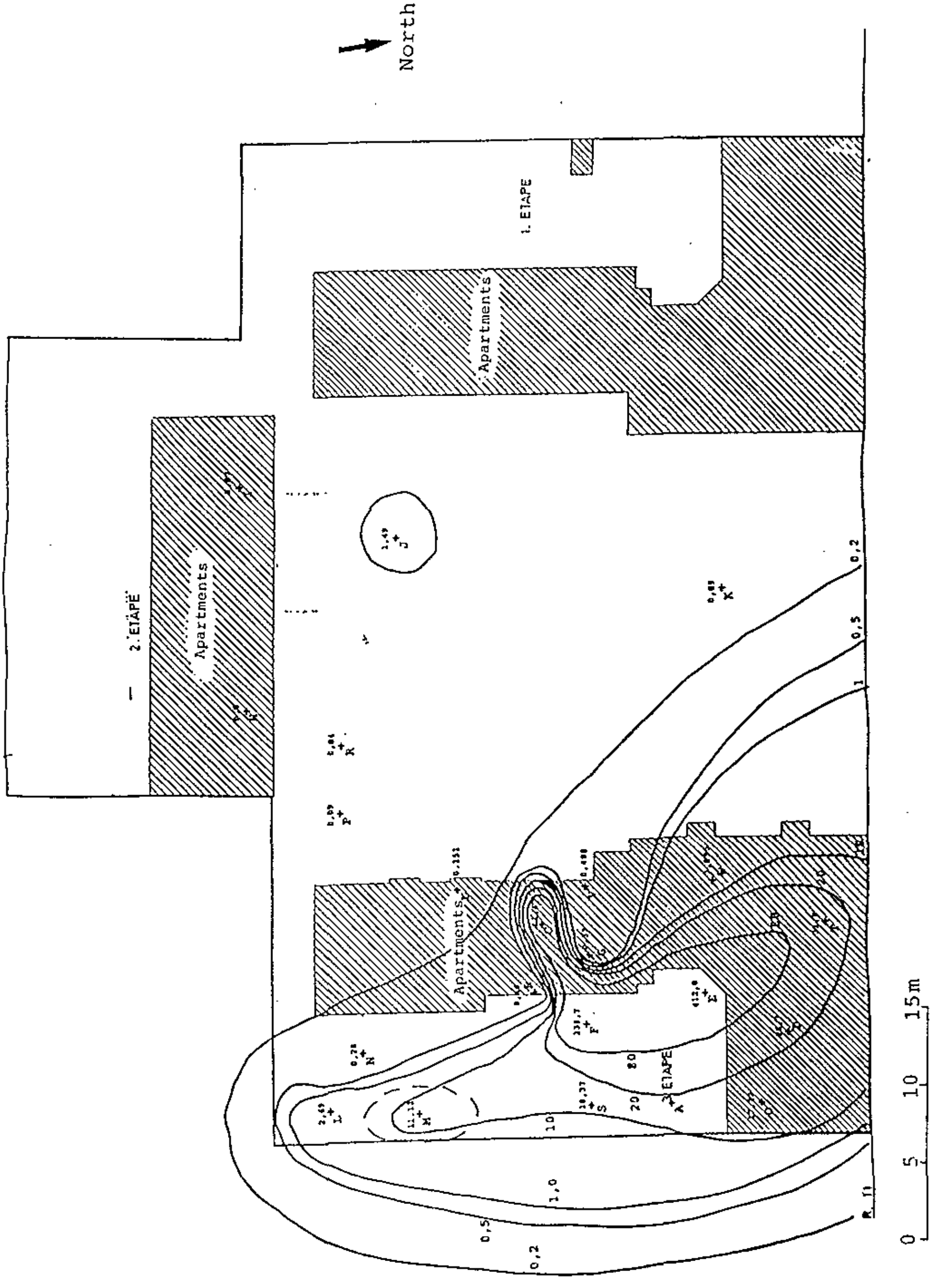
The direct measurements in the soil samples above the ground water showed high concentrations in the same places where the ground water pollution was found, as shown in figure 2. In 30% of the boreholes were high concentrations ranging from 10 to 412 mg TeCE/kg soil. These were samples from 0.5-1.0 m depth. A little more than 30% of the boreholes had 50-250 times lower concentrations in the deeper samples (1.5-2 m), most of them being those showing also the high concentrations mentioned above. Of the 15 boreholes measured in both intervals 11 had higher concentrations in 0.5-1.0 m depth than in 1.5-2.0 m.

The conclusion from these results as well as the indirect measurements presented later was, that considerable diffusion of volatiles could occur in the voids of the soil (soil air), eventually causing considerable emission to the atmosphere. The indirect measurements were therefore chosen both to supply the direct measurements in a dense network of measuring points and at the same time to empirically verify or at least indicate the order of magnitude of diffusion and emission. Cost savings on sampling and boreholes were also motivating this descision.

CONCEPTUAL FRAMEWORK

For the diffusion and eventually the emission through the soil surface soil air concentrations are important. Non stationary estimation of the emission by diffusion in soil air show, that a stationary state, given by Fick's law, will be established within 0.5 to 5 months. However, it is not possible to determine the reduction of this ideal diffusion in the soil caused by porosity, moisture, sorption or even density effect of the volatile compounds, since no data were available. Then the many models available do not help much.

Figure 2. TeCE-content in soil samples from the depth 0.5-1 meter (mg/kg)



Sorption plays a dominant role in reducing the partial pressure of the volatiles in the voids of the soils above the ground waters. Consequently it is not possible directly from the pollution concentrations in the soils to estimate the diffusion. It will depend on whether the major part of the pollution is well distributed and attached to the soils or it is present as hanging drops in dead ends pores or voids.

The partial pressure of the volatiles in solution in the ground water can roughly be estimated by Henry's law.

Both by adsorption or solution partial pressures are reduced compared with saturated vapour pressure, reducing the upward gradients and thus the diffusion and emission of the volatiles. Gravity forces tend to do the same, but is even more important to possible convective down ward movements. Solution in precipitation also will be a down ward transport mechanism. It all agrees well with the fact, that there is still considerably pollution observed in the ground.

The conclusion from all the above mentioned was, that only an indirect mapping of the soil pollution could be made by measuring the soil air concentrations in 0,75 m's and 1.5 m's from a large number of spots. The results should then in the hot spot area be related to the here already available soil concentrations, thus calibrating this so-called "sniffing measuring system". To estimate the emission through the soil surface, however, a series of direct measurements appeared necessary, since the uncertainty range of the diffusion was too large. Concentrations and emissions measurements are described in detail in the next section. An easy control measurement procedure using the in this case very sensitive photoionization detector in holes rammed to 0.3 m's depth was also carried out. The air to the detector was sucked through a piece of teflon tube. It was believed, that with the chosen procedure sufficiently accurate extension of the detection of the polluted area would be given by a dense network of measuring points at a reasonable cost since boreholes and soil sampling and analysing was saved. At the same time it was believed, that a quantitative indication of the emission levels could be found.

The most important parameter was found to be the TeCE concentration, being generally 10-100 times as high as TriCE concentrations and assumed nearly as toxic (both are suspected to be carcinogenic chemicals). Thus the gas concentration was also wanted for an assessment of the health risk at the site when inhabitated.

Since technical measures are available to avoid critical diffusion into the building through the basement, the most important issue for the measurements was to decide whether the environment outdoors was too heavily loaded with volatile compounds in the atmosphere.

Measurements were carried out in the end of 1987.

MEASUREMENTS OF SOIL AIR AND EMISSION

Different methods can be used to determine the concentration of volatile organic compounds in soil air. In this case we used small diameter steel pipes (0,008 m I.D.), from which the samples were taken. This method has recently been described by ROSENBLOOM and CARLSON (1988).

The steel pipes were cleaned with detergent, rinsed and dried and hammered down to the desired depth. The dead volumen of the pipe and sampling system was flushed for 10 minutes with a flow of 100 ml/min. In the first measurering series (sampled Nov. 27. 1987), a flow of 350-450 ml/min was sampled through an activated charcoal tube with two sections containing 100/50 mg of activated charcoal by means of a vacuum pump. The absorbent tube was connected to the steel pipe with a PTFE tube, and the sampling time was 10-50 min. As the levels of concentrations, especially for TeCE, were higher than first expected, several absorbent tubes contained more TeCE than acceptable, and a second series was performed (sampling date: Dec. 9-10., 1987). Here, the absorbent tubes contained 800/200 mg of activated charcoal, the flow was 100 ml/min, and the sampling time was 10-20 min. In all cases, the pump was calibrated on site and contained an automatic flow regulation that kept the flow constant and independant of pressure drops.

The absorbed hydrocarbons were extracted in CS_2 and analyzed for tetrachloroethene (TeCE), trichloroethene (TriCE), and turpentine (Tu) by capillary gas chromatography and flame ionization detection.

By examination of the chromatograms a compound was found in several samples which was expectedly a chlorinated hydrocarbon with a lower boiling point that TriCE and TeCE. GC/MS analysis of selected samples in the second series showed this compound to be cis-1,2-dichloroethene (cis-1,2-DiCE). Cis-1,2-DiCE was by PARSONS et al. (1984) found to be present in well water at a site contaminated with TriCE from a leaking storage tank and was concluded to be transformed from TriCE by microcosms.

The results of the soil air measurements are shown in figure 3 and 4.

Measuring point	Depth: 0,75 m			Depth: 1,50		
	TeCE mg/m^3	TriCE mg/m^3	Turpentine mg/m^3	TeCE mg/m^3	TriCE mg/m^3	Turpentine mg/m^3
1	>1000	4.7	9.8	160	1.5	1.4
2	290	4.8	<0.1	100	3	0.6
3	2.1	<0.1	<0.1	20	0.2	<0.1
4	>1000	73	<0.1	>1000	52	6.3
5	20	0.8	<0.1	26	0.3	<0.1
6	>1000	61	250	>1000	53	96
7	>1000	88	<0.1	>1000	45	1.1
8	>1000	38	<0.1	>1000	37	<0.1
9	>1000	210	2200	>1000	220	2800
10	>1000	11	0.6	>1000	18	0.5
11	>1000	57	170	>1000	200	130
12	490	58	<0.1	>1000	59	<0.1
13	420	3.3	<0.1	250	4.2	<0.1
14	5.2	<0.1	<0.1	1.8	<0.1	<0.1
15	380	26	<0.1	360*	15*	<0.1*
16	660	17	<0.1	99**	1.4**	150**
17	1000	7	0.8	>1000	18	<0.1

* Depth: 0.40 m (high ground water level)
** Depth: 1.40 m (high ground water level).

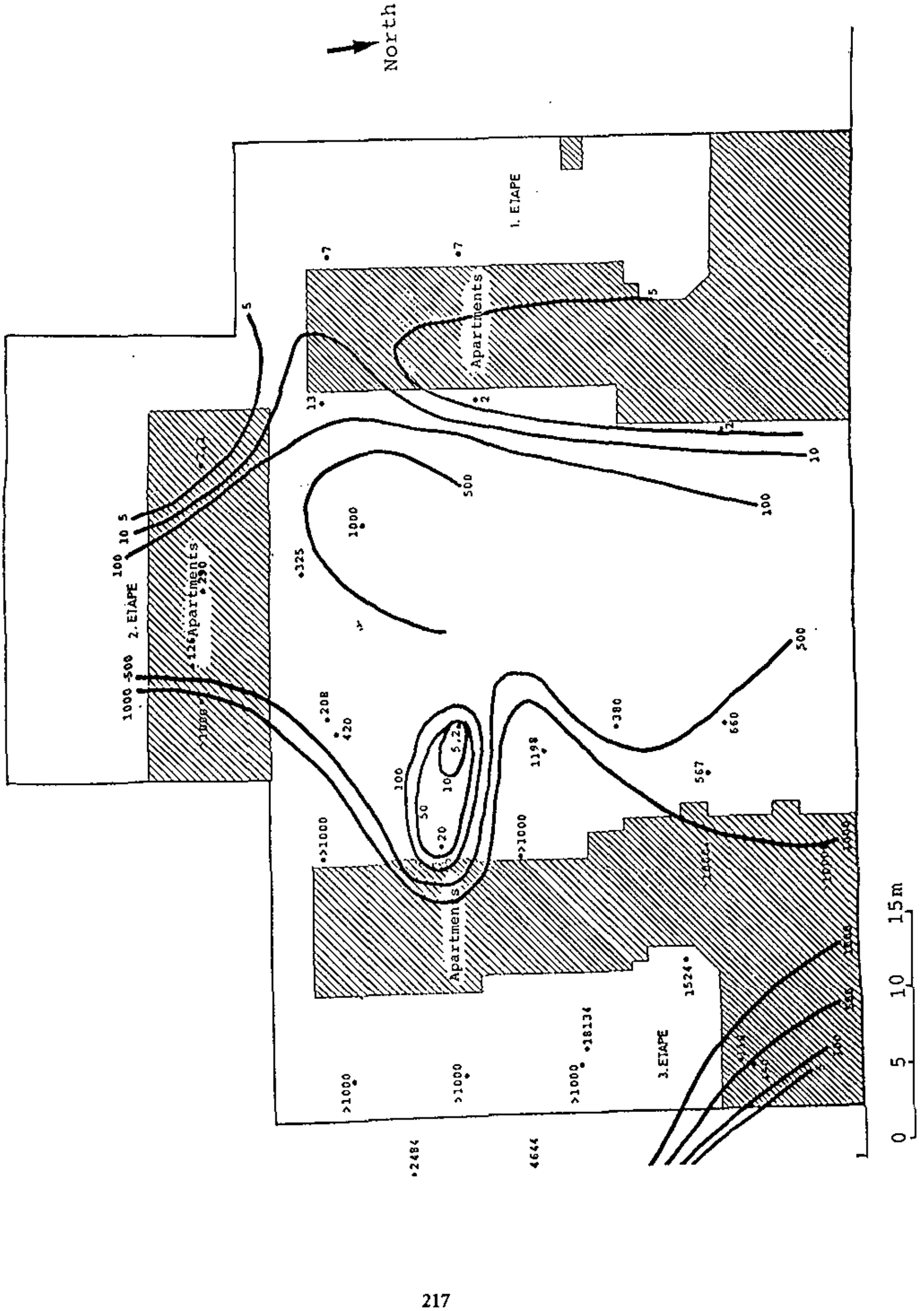
Figure 3. Results of the first series of soil air measurements, NOV. 27. 1987.

Measuring point	Depth: 0,75 m			Depth: 1,50		
	TeCE mg/m^3	TriCE mg/m^3	cis-1,2-DiCE mg/m^3	TeCE mg/m^3	TriCE mg/m^3	cis-1,2-DiCE mg/m^3
21	7	9	100	1143	559	86
22	18134	326	<3			
23	110	8	<3			
24	1524	58	19	6104	158	23
25	208	18	<3			
26	1198	78	<7			
27	567	8	<2			
28	126	7	<3			
29	325*	7*	<2*			
32	13	<2	<2	8**	<3**	<3**
33	<2	<2	<2	<3**	<3**	<2**
34	<2	<2	<2	***	***	***
35	7	<3	<2	3	<3	<2
36	7	<2	<2	7	<3	<2

* Depth: 0.64 m (due to hard soil)
** Depth: 1.70 m (1,50 m below new sand layer)
*** Excluded due to hard soil

Figure 4. Results of the second series of soil air measurements, DEC. 9-10. 1987.

Figure 5. TeCE-content in soil air samples from the depth 0.75 meter (mg/m3)



In figure 5 the results from the depth 0,75 m are shown on a map of the site. The gaseous emission of TriCE and TeCE from the soil was measured by means of a flux chamber as evaluated by DUPONT (1987). The instrumental set-up is shown in figure 6.

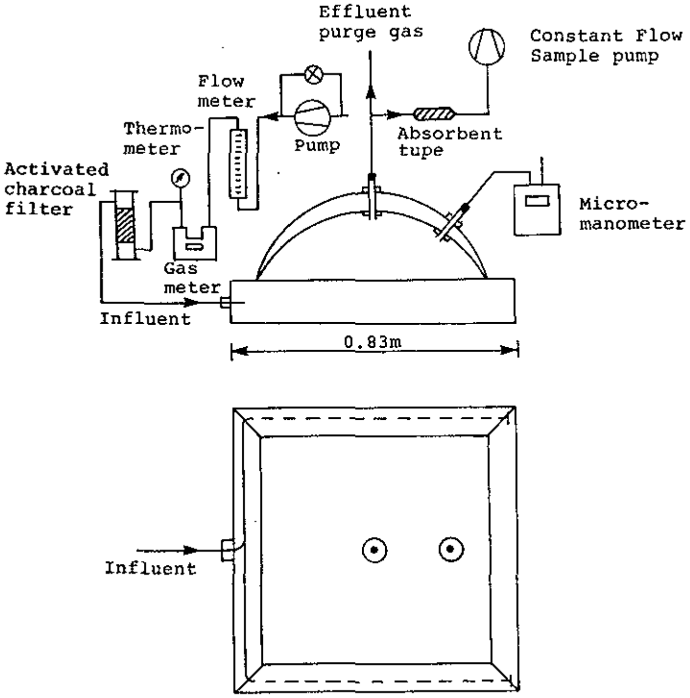


Figure 6. Instrumental set-up of flux chamber measurements

The flux chamber was covered with a layer of TEDLAR^R inside, its area was 0.83 x 0.83 meter, and the volume of the chamber was 35 liters. The flux chamber was pressed onto the soil surface, and 5 l/min charcoal-filtered purge gas was introduced into the chamber through a perforated PTFE tube tightened to the interior of the chamber to ensure good mixing conditions. From the effluent purge gas a partial volume of 2-3 l/min was sampled through 800/200 mg activated charcoal for 50-80 min. The purge and sampling system was activated in 120 min. (giving 17 times of mixing) before the absorbent was introduced. With a micromanometer it was ensured that the chamber had no leaks and that no flow variations occured. The influent flow was measured by a gas meter, and a calibrated vacuum pump with automatic flow regulation ensured the sampling flow. Background air was sampled with identical sampling equipment with a sampling flow of 1 l/min for 6 hours. Extraction and analysis was performed as described for the soil air measurements. The results of the flux chamber measurements are shown in figure 7.

Measuring point	Flux TriCE mg/m^2/h	Flux TeCE$_2$ mg/m^2/h	Flux cis-1,2-DiCE mg/m^2/h
22	0,07	5,0	< 0,02
26	2,0	50,3	0,05
27	0,04	1,1	< 0,03
33	0,02	1,1	< 0,01
Background (mg/m^3)	< 0,009	< 0,009	-

Figure 7. Flux chamber emission and background concentration measurements. Date of sampling: Dec. 9-10., 1987.

DISCUSSION

Soil sampling (for direct chemical analysis) has not occured at exactly the same places as soil air sampling, because they were made in two different investigations. This is unfortunate for more scientific purposes (empirical data on transport phenomena and equilibria). For a pollution mapping in a specific case, this weakness in the data set is not serious, as long as there is a good qualitative agreement between the results of soil concentrations and soil air concentrations, which is the case here. At least a certain ratio of samples of soil and soil air at the same spots would have helped and should be used in future cases since being easily obtainable. A more extensive series of measurements with this condition fulfilled on different types of soils should be carried out to obtain a better empirical background.

It is well known that the concentrations of water samples depends only on the water exactly at the sampling location, and this can then vary very much, because of different origin of the water from place to place. The measured soil air concentrations have in this relation bigger variations than expected. Since it was assumed, that due to the diffusion in the unsaturated zone, soil air samples would reflect pollution signals from some distance, thus being less local. Ground water fluctuations may also induce convective gas movements in the top soil increasing both horizontal and vertical dispersion of volatiles here. Apparently the air sampled depended very much on high local pollution concentrations f.i. caused by adsorption on clays, being some what similar to the water situation. Some delocalized effect is shown though (from neighbouring soil columns or the soil and ground water beneath) since the ranges and relative variations are less drastic than for soil and water samples.

The highest levels of soil air concentrations were found in an area with a representative soil concentration value of 100 mg TeCE/kg and 0.1-1 mg TriCE/kg, i.e. high, but far less than possible by filling the pore volume. The maximum soil air concentrations of volatiles were here 18 g TeCE/m^3, 0.6 g TriCE/m^3 (and 2.8 g Tu/m^3). Thus the ratio of max. actual vapour pressure is 0.13 (TeCE), 0.0014 (TriCE) and 0.085 (Tu). In other words vapour pressures in the soils never came near the saturated vapour pressures of the pollutants as free liquids. The highest pollution (TeCE), however, also shows the value nearest of to saturated vapour (0.13 i.e. 13%) in the pores.

In estimates of diffusions by soil concentrations and soil air concentration, several factors can be of importance. Diffusion can be changed by locked air pockets due to moisture, condensing waters or ice building in the pores. This can reduce the effective porosity drastically and cause a build up of higher vapour pressures temporarely. One has then to ensure, that the measurements are not taken under a transition state resulting in other diffusion conditions that actually assumed. For instance to an overestimated average impact of the atmosphere can be the result of using measurements taken in temporary elevated vapour pressure conditions, even though they may actually not exist because of a contemporary reduced porosity.

The emissions from the soils were planned to be measured by flux chamber in about 10 spots as an empirical verification of the levels estimated by conservative theoretical diffusion calculations (Fick's law). For technical reasons only 4 measurements were taken, showing rather big variations in the result, presumably due to the variations both in pollution present and in soil texture. Flux chamber measurements are also much more sensitive to the history of the site, since even small spills by f.i. transfer of polluted soils can give severe errors. The measured fluxes (F) are in figure 8 compared with estimates from Fick's law of stationary non reduced diffusion through clean soil top layer of 0.75 m.
This is the highest diffusion possible from this depths, and these sorts of calculations often lead to high results compared with reality.

Measuring point	Soil air concentration (Cf) mg/m^3	Measured emission (flux) (F) mg/m^2/h	Theoretical emission (I) mg/m^2/h	Ratio F/I
33	<2	1.1	<0.031	>35
27	567	1.1	8.8	0.13
26	1198	50.3	18.7	2.7
22	18134	5.0	283	0.018

Figure 8. Theoretical estimates and measured emission.

Even though, the ratio F/I shows in two cases a higher flux that 1! It is assumed to be caused by unobserved surface pollution at this spot or due to one of the other sources of error as mentioned above. Using this values implies, that most of the area was assumed to be critical with respect to health outdoors. Since this "overpressure" also is in conflict with the above mentioned volatile contents in the soil being much less than the saturation value, it can be assumed to be an overestimation to use this value. At the same time, if the emission was really of this size then the pollution case would solve itself within a very short time, eventually all being evaporated. Due to the uncertainty in the actual emissions and influencing factors on soil air concentrations of volatiles a socalled "hand-brake" solution had to be introduced for deciding the future activities.

The "handbrake solution" was to identify those polluted areas, which could at all provide critical concentrations with respect to health of human beings in the lowest two meters of the atmosphere for more than half a year. These could be decided directly on basis of the total pollution contents per square meter down to the ground water, the critical concentration decided by the toxicological study and the emission calculated by average wind speed to give the critical concentration. The emission is then assumed to take place evenly distributed. Thus the "handbrake solution" is a totally abstract solution with respect to physics, except that the emissions used must not be exceeding any of the limits given by the most conservative model which present the problem. This solution showed a smaller area to be of the hazardous kind than by the diffusion modelling.

CONCLUSIONS

Volatile organic compound contents of soil samples have been compared with soil air contents and described for a pollution case in soil and ground water. Direct mapping of a pollution with volatile chlorinated hydrocarbons by chemical analysis of soil and water samples are in general agreement with the indirect mapping by analysis of soil air samples of the same chemical components. Thus it is possible by this "sniffing concept" roughly to determine polluted areas and hot spots giving considerable environmental impact, especially on the atmosphere and also on ground water. However, direct measurements of the contents of pollution is still required both as a reference and to estimate the lifetimes of the pollution present. In this case say 5-10 boreholes both shallow and deep as well as a few boreholes down streams of the pollution were sufficient reference. Supplementary measuring net points can then for more than 80% be sniffing rods to shallow depths, implying reduced costs of the held measurements. It seems reasonable in 50% of the sniffing spots to measure soil air concentrations in two depths (in this case the intervals 0,5-1 m and 1,5-2 m depth) to verify whether diffusion modelling in the soil air is possible, e.g. by ideal Fickian diffusion, to estimate the emission to the atmosphere. Large variations in soil air concentrations both horizontally and especially vertically is prohibitive for good modelling based on soil air contents including boundary conditions such as the partial pressures of the volatiles. In this case a direct measurement of the emissions by a fluxchamber in strategic set of reference points (in this case 6-10) can therefore had to be used together with the diffusion modelling to determine the levels of emissions, and thus asses the health risks of the evaporating volatile compounds. Many sources of error are present.

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Remedial actions and hydrological controls

STATUS OF THE SUPERFUND INNOVATIVE TECHNOLOGY EVALUATION (SITE) PROGRAM.

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ABSTRACT

In 1986, the U.S. Environmental Protection Agency's (USEPA's) Offices of Solid Waste and Emergency Response, and Research and Development established a technology research, demonstration, and evaluation program to promote the development and use of innovative technologies. Called the Superfund Innovative Technology Evaluation, or SITE program, the program will help provide the treatment technologies recessary to implement new cleanup standards that require a greater reliance on permanent remedies at Superfund sites. The program of the Risk Reduction Engineering Laboratory has been focused along three lines: (1) a fill-scale demonstration program; (2) an emerging technologies development program; and (3) innovative technologies development within the USEPA.

The Demonstration Program (DP) represents a unique partnership between the USEPA and technology developers. Although most projects involve nu funding for developers, a mutually beneficial relationship is established for both parties. Developers are responsible for mobilizing their units and operating them at selected sites. These may be Superfund sites, developer´s facilities, USEPA laboratories, or private sites. The USEPA is responsible for sampling, analysis, and evaluation of test results. The developers are provided extensive data that validate their capabilities while the USEPA is able to assess the performance, reliability, and cost of technologies. This information will be used directly by Regional and State personnel responsible for the selection of remedies and responses at Superfund sites. Ad of July 1988, 29 technologies are in the DP.

The Emerging Technologies Program (ETP) was initiated in 1987. This pertion of the SITE program fosters the further development of technologies that are not yet ready for full-scale demonstration. The goal is to ensure that a steady stream of more permanent, cost-effective technologies will be ready to be demonstrated in the field, thereby increasing the number of viable altyernatives available for use in Superfund cleanups. In July 1988, the first seven technologies were accepted into this program.

The Innovative Technologies Program (ITP) is an outgrowth of technologies that have been primarily developed through the USEPA´s Lead. This portion af the SITE program fosters the further development and acceptance of these technologies by firms that are willing to commercialize these technologies for use at Superfund sites.

INTRODUCTION

The 1986 amendments to Superfund (1) create a comprehensive program of research, development, demonstration, and training to promote the development of alternative and innovative treatment technologies. The terms "alternative or innovative treatment technologies" are defined in the law to mean those technologies that permanently alter the composition of hazardous wastes through chemical, biological, or physical means so as to significantly reduce the toxicity, mobility, or volume. The terms also include technologies that characterize the extent and nature of site contamination and technologies that assess the stresses imposed by contaminants.

In order to carry this legislative mandate, the USEPA has established the SITE program. A strategy and program plan has been developed and published(2) and the progress and accomplishments of the program were reported to the United States Congress in February 1988.(3)

The overall goal of the SITE program is to maximize the use of alternatives to land disposal in cleaning up Superfund sites and to encourage the development and demonstration of new, innovative treatment and monitoring technologies. This paper will concentrate on the technology development and demonstratio aspects of the program.

The technology proram has been designed to accomplish the above goals along three lines: (1) a full-scale Demonstration Program; (2) an Emerging Technologies Proram (ETP); and (3) Innovative Technology Development (ITD) within the USEPA.

DEMONSTRATION PROGRAM

The purpose of the demonstration and evaluation of selected technologies is to develop performance, cost-effectiveness, and reliability data so that Superfund decision-makers can make sound judgments as to the applicability of the technology to a perticular site. The results of the demonstration should identify the limitations oof the technology, the wastes and media to which it can be applied, the operating procedures, and the approximate capital and operating costs. The demonstrations will be carried out at full-scale or at a scale that allows valid comparison and direct scale-up to commercial size units. The duration of the demonstration will be determined on a case-by-case basis but must be of a sufficient time to adequately characterize equipment reliability and operational variabilities.

ITE demonstrations usually will be conducted at actual uncontrolled hazardous waste sites including Superfuns sites, state sites, sites from other Federal agencies, and private party sites. Occasionally, demonstrations will be conducted at USEPA-operated test and evaluation (T&E) facilities. These T&E facilities would have adequate pollution control and safety equipment in place so that technologies could be evaluated over a wide range of operation without pollutant releases outside of the facility. The T&E facilities could provide for faster testing of smaller and moderate size units under safe and controlled conditions at a lower cost. The tests conducted in these facilities would be on wastes brought in from an actual hazardous waste site.

The funding of the demonstration will be cost-shared between the USEPA and the applicant. The technology developer usually will be expected to pay the costs to erect and operate the equipment on-site and dismantle and remove the equipment at the end of the demonstration. The USEPA will pay for the costs of sampling and analysis, quality assurance and quality control, evaluating data, and preparing reports. The USEPA also will help the developer obtain any required permits. Normally, there will be no exchange af funds between the Agency and the applicant. In a few instances where the technology is unique, unusually promising, and high in financial risk, the USEPA will consider bearing a greater portion of the total project cost if the developer is unable to obtain financing elsewhere.

At the completion of the demonstration, the USEPA will undertake activities to encourage wide-scale use of the successful technologies in the Superfund cleanup program. The Agency will evaluate the application of these technologies to other sites and to wastes other than those tested. A technology transfer and technical assistance program will transmit results to potential users within and outside of the Agency.

The USEPA has issued three requests for projects under the demonstration program, one in 1986, the second in January 1987, and the third in January 1988. Another request for projects will be issued in January 1989.

As of July 1988, 29 technologies have been accepted into the program. These are summarized in Table 1. Demonstrations have been completed. The Shirco Infrared System was evaluated at two Superfund sites, i.e., the Peak Oil site in Brandon, Florida, and the Rose Township site in Michigan. The Hazcon solidification/stabilization process was demonstrated at the Douglassville, Pennsylvania, Superfund site and the International Waste Technologies in situ stabilization process at a Miami, Florida, PCB site. Terra Vac´s in situ vacuum axtraction process was teted in Groveland, Massachusetts. To evaluate the American Combustion burner, waste from the Stringfellow site in California was brought to the USEPA´s Combustion Research Facilicy in Jefferson, Arkansas.

EMERGING TECHNOLOGIES PROGRAM

The purpose of the ETP is to foster the further development of technologies that are not yet ready for full-scale demonstration. The goal is to ensure that a steady stream of more permanent, cost-effective technologies will be ready to be demonstrated in the field, thereby increasing the number of viable alternatives available for use in Superfund cleanups. The ETP will deal with innovative technologies for recycling, separation, detoxification, destruction, and stabilization of hazardous chemical wastes. These emerging technologies will include, but not be limited to, chemical, biological, thermal, physical, stabilization/solidification, etc.

The ETP will provide 2-year funding to technology developers that take promising bench/laboratory-scale technologies to the pilot-scale. The ETP will pay up to $150,000 per year, for a maximum of $300,000 over 2-years, available to any individual technology developer. In order to obtain second year funding, significant progress must be made during the first year. The first solicitation for ETP occurred in September 1987. Of the 84 proposals that were submitted, seven were selected for funding. The second solicitation occurred in July 1988.

The seven technologies selected from the first solicitation are summarized in Table 2. These projects will start in August/September 1988.

INNOVATIVE TECHNOLOGIES PROGRAM

The Risk Reduction Engineering Laboratory´s research and development program has resulted in several technologies that appear to have application to solving problems found at Superfund sites. The goal of the ITP is to facilitate and accelerate the development of those technologies from the pilot-scale to full-scale demonstration and commercialization. After the USEPA has carried the technologies through the concept, development, and pilot-scale, and has performed some field evaluation, commercial firms will be sought to enter into an agreement with the USEPA to field demonstrate and commercialize the technology. In return for this commitment from the commercial sector, the USEPA will grant a user´s agreement with the firm allowing them to utilize the technology. The authority for the Federal government to enter into such an agreement is authorized by the Stevenson-Wydler Technology Transfer Act. Some of the technoligies in the ITP are listed in Table 3.

Table 1. SITE Demonstration Program Participants

Thermal Treatment

Developer	Technology	Remarks
American Combustion, Inc., Norcross, GA	Pyretron Oxygen Burner	A proprietary burner using pure oxygen in combination with air and natural gas to destroy liquid hazardous waste. The use of oxygen allows higher temperatures which improves the kinetics of waste destruction, allows higher waste throughput, and reduces stack gas volume. Solids and sludges can be coincinerated when the burners are used in conjunction with rotary kiln furnaces. The burner is capable of achieving temperatures as high as 5000°F and should effectively destroy wastes which require high temperatures for destruction (e.g., halogenated organics) or have low heating values (small amounts of organic constituents in an inert base). Demonstration is completed.
Haztech/EPA Region 4, Atlanta, GA	Shirco Infrared Thermal Destruction	A mobile thermal destruction process uses rows of electrically powered silicon carbide rods to bring the waste to combustion temperatures. Remaining combustibles are destroyed in an afterburner. The full-scale, four-component system can process 100 to 250 tons of waste per day, depending on the waste characteristics. The first component, the primary furnace, is lined with layers of lightweight ceramic fiber blanket insulation. The furnace generates temperatures up to 1,850°F by using infrared radiant heat provided by horizontal rows of silicon carbide rods (located above the conveyor belt). Waste moves through the primary furnace on a woven wire mesh belt. The second component, an infrared or gas-fired secondary combustion chamber, is capable of reaching temperatures up to 2,300°F. The secondary chamber destroys gaseous volatiles from the primary furnace. The third component consists of an emissions control system that removes particulates in a venturi scrubber. Acid vapors are neutralized in a packed tower scrubber, and an induced draft blower draws cleaned gases from the scrubber into the exhaust stack. Demonstration is completed.
Shirco Infrared Systems, Inc., Dallas, TX	Infrared Thermal Destruction	A portable pilot (one ton per day) Shirco unit was evaluated at the Rose Township-Demode Road Superfund site. Demonstration is completed.

Table 1. SITE Demonstration Program Participants (continued)

Developer	Technology	Remarks
Ogden Environmental Services, San Diego, CA	Circulating Fluidized Bed Combustor	A circulating fluidized bed combustor. This combustor operates at a higher velocity airflow and produces a much higher combustion efficiency than a conventional fluidized bed. High turbulence in the combustor allows the process to destroy a wide variety of waste materials at temperatures below 1,560°F (850°C). The unit employs simultaneous limestone injection that captures the acid gases and eliminates the need for a scrubber. To be demonstrated in 1988.
Retech, Inc., Ukiah, CA	Plasma Heat	This technology uses a centrifugal reactor with plasma heat to decompose organics in a mixed solid and liquid feed. The solid components are melted and cast or granulated for disposal. The volatile compounds are vaporized and decomposed in an afterburner also heated by plasma heat. Off-gases are also treated conventionally. To be demonstrated in 1988.
Toxic Treatments, Inc., San Mateo, CA	In Situ Steam/ Air Stripping	Steam and air are injected into the soil through a series of drill holes to strip volatile contaminants. To be demonstrated in 1989.
Westinghouse Electric Corporation, Madison, PA	Pyroplasma System	The pyroplasma process pyrolyzes wastes using a thermal plasma field. The heart of the system is a plasma arc torch which produces a thermal plasma with temperatures of more than 9,000°F. Waste liquids are injected directly into the plasma where the molecules are broken into their atomic states and the atoms then recombine to produce hydrogen, carbon monoxide, nitrogen, hydrogen chloride particulate carbon, carbon dioxide, ethylene, and acetylene. The product gas is scrubbed with caustic soda to neutralize and remove the acid gases and remove particulates. To be demonstrated in 1988.
		Chemical Treatment
CF Systems Corporation, Cambridge, MA	Solvent Extraction	Solvent extraction using liquified gases near their critical conditions as solvents to remove organic pollutants from sludges, solids, or liquid waste. To be demonstrated in 1988.

Table 1. SITE Demonstration Program Participants (continued)

Developer	Technology	Remarks
Resources Conservation Company, Bellevue, WA	Solvent Extraction	The B.E.S.T. extraction sludge treatment process was developed to dewater and deoil contaminated sludges and soils, including those containing PCB's. A chemical plant-like process uses differences in chemical miscibility at different temperatures to break waste down into three distinct fractions: (1) dischargeable water, (2) reusable oil and organics, and (3) dry, oil-free solids. The transportable unit treats solid particles no larger than 1/4 of an inch. To be demonstrated in 1989.
Ultrox International, Santa Ana, CA	Ultraviolet Radiation and Ozone	Treatment of organics in groundwater, and leachate by ultraviolet radiation and ozone. To be demonstrated in 1989.
		Biological Treatment
Air Products and Chemicals, Inc., Allentown, PA	Fluid Bed Biological Systems	Mobile fixed-film, fluidized bed unit to treat aqueous waste with organic substances. Pure oxygen used. To be demonstrated in 1989.
Biotrol, Inc., Chaska, MN	Fixed-Film Plug Flow Bioreactor	Uses amended microbial consortium to treat organics in groundwater and leachates in a fixed-film plug flow reactor. To be demonstrated in 1989.
DETOX Industries, Inc., Sugarland, TX	Biological Degradation	Proprietary nationally-occurring organisms are utilized in a reactor in a batch process to degrade organic contaminants. To be demonstrated in 1988.
DETOX, Inc., Newport Beach, CA	Fixed-Film Bioreactor	Biological treatment of liquids with low organic concentrations. The fixed-film bioreactor should result in short retention times. To be demonstrated in 1989.
MoTec, Inc., Mt. Juliet, TN	Liquid/Solid Contact Digestion	Portable high energy mixer for biodegradation of organic waste in soils and sludges. To be demonstrated in 1989.
Zimpro Environmental Control Systems, Rothschild, WI	Powdered Activated Carbon/ Biological	Combination of biological treatment, powdered activated carbon-treatment, and wet air oxidation for treatment of concentrated aqueous waste. To be demonstrated in 1989.

Table 1. SITE Demonstration Program Participants (continued)

Physical Treatment

Developer	Technology	Remarks
Biotrol, Inc., Chaska, MN	Soil Washing	Physical size separation and water washing process for treating contaminated soils. To be demonstrated in 1989.
CBI Freeze Technologies, Inc., Plainfield, IL	Volume Reduction by Freezing	Physical separation and concentration of contaminants from water using a freezing system mounted in a trailer. To be demonstrated in 1989.
E.I. Dupont de Nemours, Newark, DE	Microfiltration	Treatment of liquid waste having 10 ppm to 30 percent solids with high-density polyethylene filter media. To be demonstrated in 1989.
Sanitech, Inc., Twinsburg, OH	Ion Exchange	Use of ion-exchange-like materials to selectively remove heavy metals from water. To be demonstrated in 1989.
Terra Vac, Inc., Dorado, PR	In-Situ Vacuum Extraction	Volatile contaminants are vacuum extracted from the soil through wells and removed from the air stream by carbon. Demonstration is completed.

Solidification/Stabilization

Developer	Technology	Remarks
Geosafe Corporation, Richland, WA	In-Situ Vitrification	Electrodes inserted into the ground to heat the surrounding soil to 2,000°C which melts the soil. A hood collects the off-gases which are treated. A chemically inert, crystalline product is produced from the soil. To be demonstrated in 1989.
Chemfix Technologies, Inc., Metairie, LA	Chemical Fixation/ Stabilization	Proprietary process that stabilizes high molecular-weight organic and inorganic constituents in a waste slurry using soluble silicates, silicate setting agents, and additives to crosslink with waste components to produce a stable, solid matrix. To be demonstrated in 1989.
Hazcon, Inc., Katy, TX	Solidification/ Stabilization	Process blends contaminated soil or sludge with cement, pozzolana, and a proprietary ingredient called Chloranan which aids in the solidification of organics. System is truck-mounted. Demonstration is completed.

Table 1. SITE Demonstration Program Participants (continued)

Developer	Technology	Remarks
International Waste Technologies, Wichita, KS	In-Situ Stabilization	Deep soil mixing equipment is used to drill and blend a proprietary solidification/stabilization ingredient in situ. Drill is 0.9m in diameter and injects the slurry additive from the bottom of the shaft. Demonstration is completed.
Separation and Recovery Systems, Irvine, CA	Lime-Based Technology	Lime and other minor chemicals are specially prepared to improve their reactivity to fix sludges with high concentrations (up to 3 percent) of hydrocarbons. Metals are also immobilized. To be demonstrated in 1989.
Silicate Technology Corporation, Scottsdale, AZ	Silicate Treatment	Contaminated material is separated by size. The coarse material is shredded and sized and mixed with the fines for the solidification process. Proprietary reagent is added in a batch plant and conveyed to a mixing truck. To be demonstrated in 1989.
Soliditech, Inc., Houston, TX	Solidification	Proprietary reagent is added to the slurried hazardous material in a batch process. Reagent is microblended with the waste and microencapsulates the hazardous compounds by crosslinking organic and inorganic particles, coating larger particles, and sealing small pores and spaces. To be demonstrated in 1989.
Waste Chem Corporation, Paramus, NJ	Volume Reduction/ Solidification	A mobile volume reduction and solidification system designed to decontaminate and solidify liquids, sludges, and soils. A fluidized waste stream and asphalt are mixed in a heated screw mixer system. The organics that are volatilized are treated via ozonation and carbon absorption and the waste-asphalt mixture is discharged into drums where it solidifies. To be demonstrated in 1989.

Table 2. SITE Emerging Technologies Program Participants

Developer	Technology	Remarks
Atomic Energy of Canada, Ltd., Ontario, Canada	Toxic Metals Removal	The use of ultrafiltration in combination with water soluble macromolecular compounds to selectively remove heavy metal ions from aqueous waste solutions.
Battelle Memorial Institute, Columbus, OH	Electroacoustic Soil Decontamination	The in situ treatment of soils contaminated with fuel oil, hazardous organic compounds, and heavy metals by the application of electric and acoustic fields. The electroacoustic process will be used to remove the contaminants from the capillaries.
Bio-Recovery Systems, Inc., Las Cruces, NM	Sorption of Heavy Metals by Alga SORB	Immobilized algae cells in a silicon gel polymer are used to form ion-exchange-like resins to remove heavy metal ions from aqueous solutions.
Colorado School of Mines, Golden, CO	Wetlands Treatment to Remove Heavy Metals	A wetland-based treatment technology is based on the concept of using national geochemical and biological processes inherent in a wetland ecosystem to remove and accumulate metals from influent waters. A man-made wetlands will be evaluated.
Energy & Environmental Engineering, Inc., Somerville, MA	Laser Stimulated Photochemical Oxidation	Suspended solids are removed from groundwater and washed. The wash water and groundwater are then treated in a photochemical reactor utilizing a laser beam.
Envirite Field Services, Inc., Atlanta, GA	Solvent Soil Washing	Soil is washed utilizing a potential blend of solvents. The solvent is removed from the soil by steam stripping.
Western Research Institute, Laramie, WY	In Situ Oil Recovery and Biodegradation	Oil and waste are recovered from the soil by controlled injection of steam, and hot and cold water. Residual organic pollutants in the soil are biodegraded.

Table 3. SITE Innovative Technologies Program

Technology	Remarks
Mobile Soil Washer	A mobile soil washer has been developed. Soil washing separates contaminants from soil by high energy mixing of soils and solvents, additives, surfactants, acids, and bases. The soil washer consists of three components: a drum washer, a countercurrent extraction chamber, and a dewatering unit. Turbine mixers and air agitators keep the solution well mixed and efficiently strips out highly volatile organics. A vapor collection system removes volatile organics to carbon treatment units. The drum washer has a maximum throughput capacity of 18 cubic yards per hour, and the countercurrent trailer is limited to two cubic yards per hour.
KPEG Treatment	Potassium polyethylene glycolate reagents are effective dehalogenators of aromatic and aliphatic organic materials, including PCB's and other toxic halides. The KPEG reagent reacts with the chlorine atoms in the aryl ring of halogenated aromatic contaminants to produce innocuous ether and potassium chloride salt. In some KPEG reagent formulations, dimethylsulfoxide is added as a co-solvent to enhance reaction rate kinetics. KPEG reagents are stable in air, tolerate moisture, are easily stored, and can be safely transported to problem sites unlike conventional anaerobic dehalogenating reagents. A large portable KPEG reactor (400 gallons) has been demonstrated on PCB-contaminated soils and a smaller pilot unit on oily pesticide wastes and liquid wood-preserving wastes.
In Situ Containment/ Treatment System	A mobile in situ containment/treatment system has been developed. The system is mounted on a 43-foot drop deck trailer and includes a diesel electric generator, an air compressor, mixing tanks, hoses, a solids feed conveyor, pipe injectors, soil testing apparatus, and accessory items. In situ containment and treatment is accomplished by direct injection of grouting material into the soil around the contaminated area in order to isolate the spill. The hazardous materials are then treated in place by oxidation/reduction, neutralization, or precipitation. When necessary, contaminated water can be withdrawn from wet wells and treated by other means.
Spent Activated Carbon Regenerator	In order to provide a safe and effective method for handling contaminated carbon, a mobile unit for detoxifying/regenerating contaminated carbon at the cleanup site has been developed. The mobile detoxification/ regeneration system, mounted on a 45-foot long semitrailer, is equipped with a rotary kiln, a second stage combustion chamber, and a gas scrubber. The unit is self-contained except for fuel and water supplies.

Table 3. SITE Innovative Technologies Program (continued)

Technology	Remarks
Electrokinetic Removal of Contaminants from the Ground	The use of an electrokinetic process, in conjunction with pumping, to expedite ion migration and removal from a saturated soil system. A series of wells is used as anodes and cathodes across which a direct current is applied. The current density results in an accelerated movement of charged ions. The effect of ion migration is greater with pulse pumping.

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A DECONTAMINATION STRATEGY FOR AN ORGANICALLY POLLUTED AQUIFER

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Summary

The glacial outwash aquifer underlying the Gloucester Landfill near Ottawa, Canada, has become polluted with toxic organic chemicals following the disposal of laboratory solvents in shallow trenches immediately above the aquifer. The remedial options considered by the environmental authorities for decontaminating the aquifer are discussed. Impermeable barrier walls were rejected as being unsuitable given the permeable nature of the underlying bedrock. It is argued that it is improbable there are existing pools of liquid organic chemicals (DNAPLs) remaining within the aquifer, therefore, much of the plume can be removed hydraulically over a period of 4 years by the operation of four purge wells

pumping to an on-site treatment plant from which the purified water is returned to the aquifer by recharge wells. The residual contamination could be cleaned up by in situ biorestoration techniques presently under development.

INTRODUCTION

The Gloucester Landfill is situated on top of a sequence of fluvioglacial and littoral deposits of Quaternary age and Paleozoic limestone bedrock. It is located on the property of Ottawa Internationl Airport (75°38'W, 45°18"N), 10 km south of downtown Ottawa, Ontario, Canada. Between 1969 and 1980, various government agencies disposed liquid wastes in a "Special Waste Compound" (SWC) at one corner of the Gloucester municipal landfill (see Figure 1). The wastes were predominantly organic solvents in glass containers from various laboratories in the nearby city of Ottawa. The bottles were placed in trenches and combusted by detonation of explosives set within the wastes.

By 1982, it was apparent that this method of waste disposal was causing the pollution of the outwash aquifer, shown in Figure 2, underlying the SWC. This aquifer is a thick (20-25 m) sequence of interstratified silts, sands and gravels believed to have been deposited as coalescing and overlapping subaqueous esker fans that formed beneath standing water adjacent to a glacial ice front. It is confined by marine silts which form a discontinuous aquitard and which in turn is overlain by an unconfined sand aquifer (Jackson et al., 1985).

The outwash aquifer has an average hydraulic conductivity of 5 x 10^{-4} m/s, while the mean transmissivity is 9.8 x 10^{-3} m^2/s and the storativity is of the order of 2 x 10^{-3}. The average groundwater velocity in the aquifer is approximately 7 cm/day (σ = 5 cm/day) as measured by borehole dilution techniques (Jackson et al., 1985) and by tritium measurements (Michel et al., 1984). The organic carbon content of the outwash aquifer materials varies from 0.1% by weight to 0.6%, with a mean of 0.35% (Jackson et al., 1985). The outwash aquifer materials are principally quartz and feldspar with minor amounts (i.e. <10%) of mica, calcite, dolomite, hornblende and garnet.

CONTAMINANT MIGRATION AND FATE

The patterns of groundwater contamination in the outwash aquifer (see Figure 2) due to these disposals suggest that the organic solvents have undergone chromatographic dispersion, i.e., aqueous-phase solute transport with sorption of the contaminants, principally by organic carbon particles within the aquifer sediments (Patterson et al., 1985). The organic contaminants have been transported distances that are inversely proportional to their respective octanol-water partition coefficients. Since the octanol-water partition coefficient (K_{OW}) provides a measure of a compound's propensity to become sorbed by organic matter (i.e., hydrophobic partitioning), the transport of the organic solvents by groundwater is in accord with the hydrophobic sorption theory of Schwarzenbach and Westall (1981) and others.

Recent sampling of the contaminated groundwater shows that degradation products from the transformation of chlorinated aliphatic hydrocarbons have appeared in the plume of organically polluted groundwater. These include vinyl chloride and 1,1-dichloroethene (see Table 1), which are produced by abiotic and biotic dechlorination reactions as shown in Figure 3.

Parameter	Abbreviation	67M-9	54M-15	128P
Vinyl Chloride	VC	15	n.d.	0.6
Chloroethane	CA	n.d.	n.d.	n.d.
1,1-Dichloroethene	11-DCE	49	66	2.1
trans 1,2-Dichloroethene	TRANS	n.d.	n.d.	n.d.
cis 1,2-Dichloroethene	CIS	1.5	n.d.	0.5
1,1-Dichloroethane	11-DCA	81	9.5	1.9
Trichloroethene	TCE	490	n.d.	n.d.
1,1,1-Trichloroethane	TCA	82	93	n.d.
Tetrachloroethene	PCE	2.9	n.d.	n.d.

Table 1. Groundwater quality data (in µg/L) at selected multilevel samplers and piezometers within the outwash aquifer, September 1987. See Figure 1 for locations. n.d. = not detectable, i.e., <1 µg/L.

In the most heavily contaminated zones of the aquifer, such as the zone of >100 µg total organo-chlorine compounds (TOCl)/L shown in Figure 2, the pH varies between 7 and 8, the E_H <200 mV and dissolved oxygen is not detectable.

AQUIFER DECONTAMINATION

Following the assessment of groundwater contamination in the outwash aquifer beneath the Gloucester Landfill, attention turned to the options available for containing the contamination or decontaminating the aquifer. A common solution that was proposed was to install an impermeable slurry wall or grout curtain around the zone of highest contamination. However, such a method presupposes that there is an impermeable floor beneath the contamination; the limestone bedrock underlying the plume is very permeable and is hydraulically connected to the overlying outwash aquifer. Furthermore, there remain uncertainties pertaining to the viability and integrity of such engineered barriers in the presence of hydrocarbons, which have been shown to increase the hydraulic conductivity of natural clays (e.g., Fernandez and Quigley, 1985).

Consequently it has been decided to decontaminate the aquifer by a "pump and treat" scheme which, it is estimated, will need to operate 4 years (Graham Engineering Consultants, 1985) and involve 4 purge wells situated along the length of the plume pumping into a treatment plant, comprising lime addition, air stripping and granular activated carbon, and 4 recharge wells to dispose of the treated groundwater. The four purge wells are to operate at a total rate of 1050 m^3/day beginning in 1990. The cost of this operation is estimated to be of the order of C$6 million (US$5 million).

In any pump and treat operation it is first essential to remove any immiscible organic liquids from the aquifer. Such dense, non-aqueous phase liquids (DNAPLs) released by waste disposal operations or by spills tend to migrate to depth leaving residual droplets of liquid within the unsaturated and saturated zones and, if present in sufficient volume, forming pools on relatively impermeable strata within the aquifer or at its base (see Schwille, 1984 and 1988). The dissolution of such residual droplets and pools can result in widespread aquifer contamination if they are not removed from the aquifer and, furthermore, could result in the recontamination of a supposedly decontaminated aquifer once the "pump and treat" operations are suspended.

There has been no unequivocal evidence of DNAPLs in the Gloucester aquifer. Inspection of shallow cores and soils removed by backhoe failed to indicate any separate organic phases. Furthermore, chemical concentrations from two multilevel monitors, both situated within fifty metres of the disposal pits, indicated that maximum concentrations of these chemicals never exceeded 4% of their solubility in the case of PCE or 1% in the cases of chloroform, benzene and carbon tetrachloride which were disposed en masse in May 1978. Finally, the vertical distribution of contamination, as depicted by the multilevel samplers used to monitor the aquifer, showed no evidence of an increase in contamination with depth, which suggests the absence of deep pools of DNAPL within the aquifer.

In order to prove the efficacy of decontaminating the outwash aquifer by purge wells (i.e., the "pump and treat" option), a test was conducted (Whiffin and Bahr, 1985) involving the injection of uncontaminated groundwater into the contaminated aquifer via an injection well and the withdrawal of a similar volume of contaminated groundwater from another well 5 meters away. In between the two wells was situated a multilevel sampler, M1, from which samples were analysed for two injected, nonreactive tracers and for three organic solvents present in that part of the aquifer -- 1,4-dioxane, tetrahydrofuran and diethyl ether. The behavior of the organic contaminants during desorption is shown in Figure 4, in which the time axis is replaced by the number of pore volumes pumped through the test section of the aquifer.

Analysis of these disappearance curves indicates that there is a log-linear relationship (Figure 5) between the number of pore volumes withdrawn from the aquifer in obtaining 90% apparent desorption of each of the organics and their respective K_{OW} values. This site specific relationship can be extrapolated to obtain an estimate of the number of pore volumes required to decontaminate the aquifer of other contaminants not found in that particular part of the contaminated aquifer which was tested in this manner.

The next step is the optimization of the placement of the purge wells so that a minimum amount of uncontaminated groundwater is pumped and sent for treatment over the specified duration of purging, by the end of which time the aquifer should have been decontaminated to a prescribed level. This is being accomplished by developing a three-dimensional (3D) groundwater flow model of the aquifer which provides reliable information for the development of simpler and less costly 2D models of groundwater flow and solute transport. The 2D groundwater flow code is embedded in the USGS code AQMAN (Lefkoff and Gorelick, 1987), which combines the flow simulation with mathematical optimization to develop and evaluate aquifer management strategies. AQMAN prepares a data file that defines the objective and all constraint functions plus other information required; the file is then used in a mathematical programming code such as MINOS (Murtagh and Saunders, 1983). The 2D code is also used to prepare input files for the simulation of contaminant transport by the USGS code MOC (Konikow and Bredehoeft, 1978), which uses the method of characteristics to solve the solute transport equation.

Because of the high cost of "pump and treat" operations, there is much interest in developing low cost alternatives, in particular ones involving in situ biological remediation. This technique, generally known as biorestoration, depends upon the ability of native microorganisms to degrade toxic organic chemicals. As pointed out by Wilson et al. (1986), many cases of organic contamination are naturally remediated without the occurrence of contamination being noticeable.

The process involves the prior removal of DNAPL pools and the subsequent injection of oxygen and nutrients into a plume of dissolved toxic organic chemicals so as to stimulate the native microbial populations to metabolize or cometabolize the organic contaminants (Wilson et al., 1986; Lee et al., 1988).

Because in situ biorestoration is still under development, it was not considered as a remedial option for the Gloucester aquifer. However, it is anticipated, that at the end of the four years of purging the contaminated aquifer, in situ biorestoration will be required to clean up the residual contamination that will undoubtedly remain in the aquifer in a cost effective manner.

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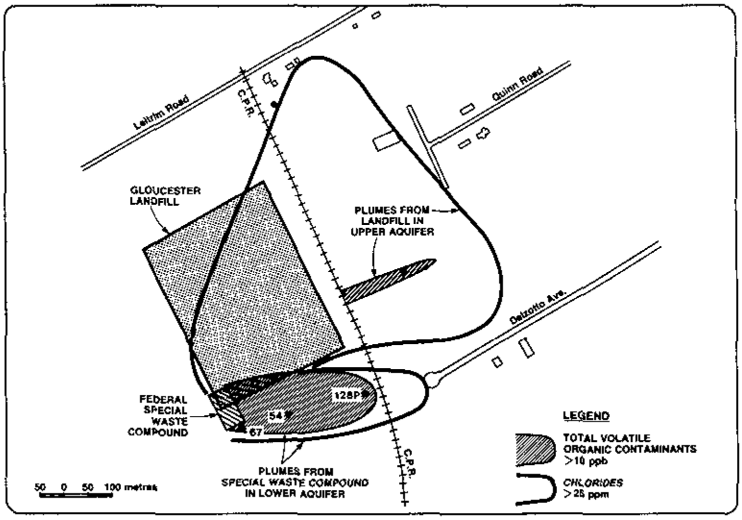

Figure 1. The Gloucester Landfill and Special Waste Compound near Ottawa, Ontario Canada

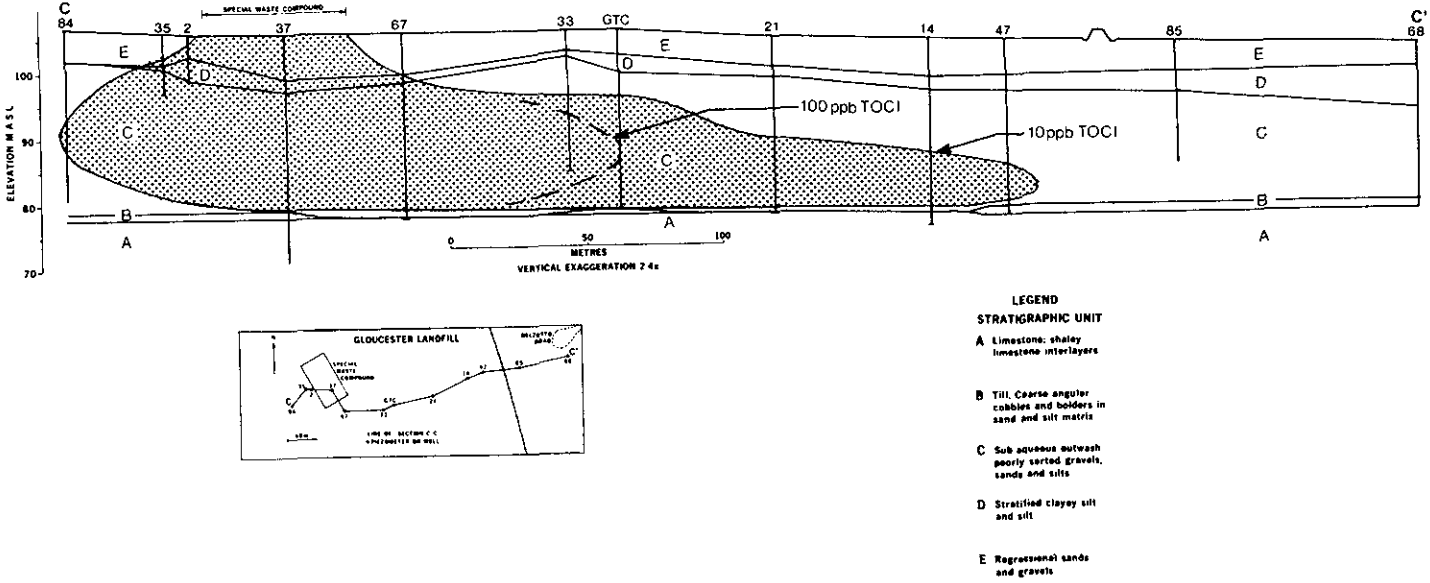

Figure 2. Hydrostratigraphy and zones of ground water contamination within the outwash aquifer.

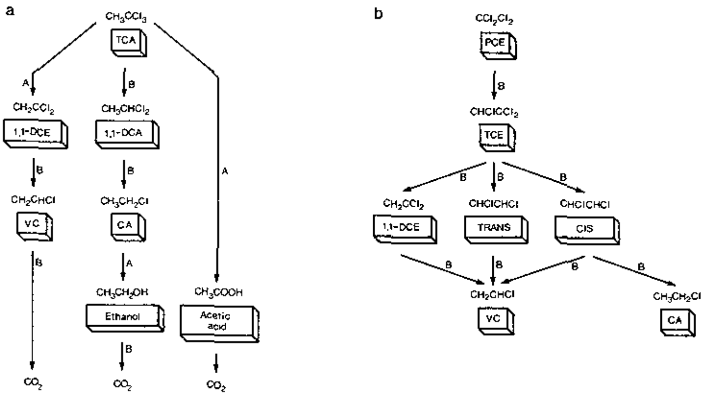

Figure 3. (a) Sequential transformation of 1,1,1-trichloroethane to vinyl chloride (VC), chloroethane (CA) and other degradation products by abiotic (labelled a) and biotic (labelled b) mechanisms (from Vogel and McCarty, 1987); (b) sequential transformation of perchloroethylene (PCE), to trichloroethylene (TCE) and to other degradation products (from Barrio-Lage et al., 1986).

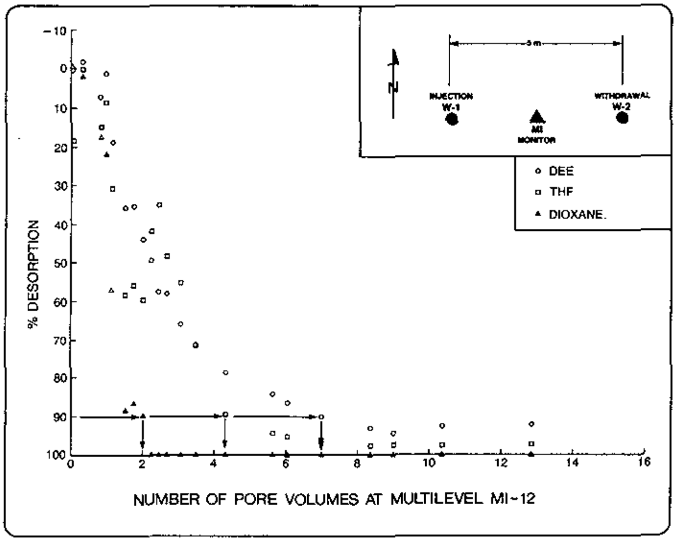

Figure 4. Behavior of diethyl ether (DEE), tetrahydrofuran (THE) and 1,4-dioxane during the purge well test.

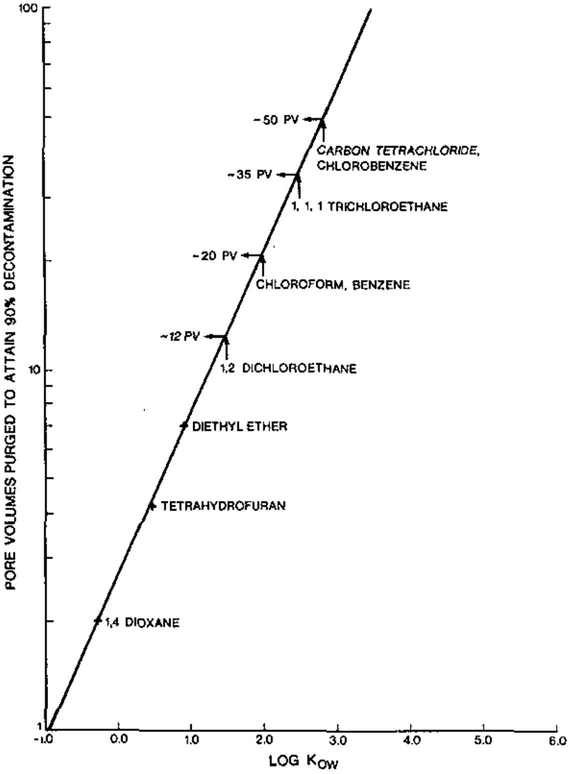

Figure 5. Estimation of the number of pore volumes that must be purged from the outwash aquifer to achieve 90% decontamination. Pore volume values for diethyl ether, tetrahydrofuran and dioxane are obtained from Figure 4.

NATO/CCMS PILOT STUDY ON DEMONSTRATION OF REMEDIAL ACTION TECHNOLOGIES FOR CONTAMINATED LAND AND GROUNDWATER - AUGUST 1988 UPDATE OF ACTIVITIES

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1. INTRODUCTION

The purpose of the NATO/Committee on the Challenges of Modern Society (NATO/CCMS) Pilot Study, "Demonstration of Remedial Action Technologies for Contaminated Land and Groundwater," is to promote the exchange of information on new and existing technologies for dealing with "problem" hazardous waste sites. The Pilot Study is based on periodic review by an international group of experts drawn from the participating countries of the results of demonstration projects carried out in a number of the participating countries. The study, which is planned to last 5 years, is "copiloted" by the Federal Republic of Germany (FRG) and The Netherlands.

This is the third in a series of articles which will publish the progress of the study on an annual or more frequent basis. The first article in the series (1) was "NATO/CCMS Pilot Study on Demonstration of Remedial Action Technologies for Contaminated Land and Groundwater." The second article (2) was "NATO/CCMS Pilot Study on Demonstration of Remedial Action Technologies for Contaminated Land and Groundwater - 1988 Activities."

The study complements and enhances programs to develop, evaluate, and demonstrate cleanup technologies in a number of the participant countries, in particular, the USA (the SITE program (3)), The Netherlands, and the FRG.

2. PROGRESS TO DATE

Representatives of six countries attended the first Workshop held in Karlsruhe, FRG, in March 1985, at which time 12 initial projects were selected for inclusion in the study (4). These projects are listed in Table 1 and are briefly described below. As the various projects are completed, further projects will be selected for inclusion in the program. Progress of the various projects was discussed at the first full technical meeting held in Washington, D.C., in November 1987 (5). A second workshop was held in Hamburg, FRG, in April 1988 (6). To date, 12 countries (Canada, Denmark, FRG, France, Greece, Italy, Japan, The Netherlands, Norway, Spain, United Kingdom, and the United States) have associated themselves with the Pilot Study in some way, although only seven (Table 1) have involved themselves to the extent of nominating demonstration projects for inclusion in the Study. At the April 1988 meeting in Hamburg, FRG, a number of new sites were submitted for inclusion in the study (Table 2). National contacts are listed in Table 3.

Each year CCMS awards a number of Fellowships for work related to the ongoing Pilot Studies. Six Fellowships were awarded in 1987 for work related to site cleanup. Five additional Fellowships were awarded in 1988.

The activities of these Fellows are meant to augment the primary objectives of the Pilot Study. It is intended that each Fellow will publish the results of his activities on a periodic basis, as data warrants, but not less than once during the life of the project.

3. THE INDIVIDUAL PROJECTS

New projects and the status of currently active projects will be reported on at the second international meeting to be held in Bilthoven, The Netherlands, on November 7-11, 1988.

3.1 Ville Mercier, Canada

The Ville Mercier is located on the south shore of the St. Lawrence River about 20 km southwest of Montreal (7). Groundwater is a major source of potable water for the local population. From 1968 to 1972, 40,000 m^3 of waste oils and liquid industrial wastes from chemical and petrochemical industries in the Montreal area were dumped into a lagoon in an old gravel pit located several kilometers southeast of Ville Mercier. In October 1971, it was discovered that several wells in the vicinity were contaminated. The site was closed in 1972, and it was estimated that 20,000 m^3 of wastes remained in the lagoons. Sampling conducted by the Quebec Ministry of the Environment (MENVIQ) revealed the presence of phenols and chlorinated organic compounds in the groundwater in the area.

A program to rehabilitate the aquifer was started by the government of Quebec in 1981. The liquid material stored in the lagoon was first removed (incineration, landfill). Following this, MENVIQ awarded contracts to develop a purge-well system and to design and construct a groundwater treatment facility.

The treatment system includes the following elements: air stripping, alum and polymer-activated flocculation-sedimentation, rapid sand filtration, and activated carbon adsorption. The sludge is disposed to a landfill. Several problems encountered were proliferation of bacteria, plugging of the treatment equipment, loss of activated carbon, and clogging of the wells. The addition of chlorine and peroxide at the head of the treatment system, and of chlorine dioxide before filtration, has solved most of these problems and improved treatment performance. The water treatment objectives have been met, except for 1,2 dichlorethane, which resists treatment because the initial concentration is very high. After 3 years' operation and pumping out of 3.6 M m^3 of contaminated groundwater, a substantial abatement in the organic contaminants has been achieved and an acceptable degree of decontamination seems likely to be achieved more rapidly than at first anticipated. It is expected that a further 3.8 M m^3 will be treated during the final 2 years planned operation of the plant.

3.2 Skrydstrup, Denmark

The Skrydstrup site is a former gravel pit used from 1963-1974 for the disposal of chemical/industrial waste from a refrigerator factory (8). Some of this waste was stored in several hundred drums at the site. There have been extensive chemical, geological, and hydrogeological investigations since 1986. Thichloroethane, trichloroethylene, and paint and acid wastes have resulted in groundwater pollution. Contaminants include haloalkyl phosphates used as flame retardants in insulating foams. The remedial action consists of digging up, removal, and offsite disposal of several hundred drums, followed by groundwater pumping and treating, and on-site aerobic/anaerobic treatment of the polluted soil by a variety of means including microbial treatment. The groundwater system will involve air-stripping and active carbon adsorption over an anticipated 10-15 years from 1988.

Drums and obviously contaminated soil were excavated and removed from the site. The following full-scale development projects have been started to deal with the contaminated soil and groundwater.

a. biodegradation of chlorinated solvents in contaminated soil;

b. aerobic biodegradation by addition of natural gas in columns with activated carbon adsorption;

c. aerobic biodegradation of chlorinated solvents in the unsaturated zone by co-metabolism by oxidation of methane and/or propane;

d. anaerobic biodegradation in the contaminated zone by addition of sodium acetate.

3.3 Charlottenberg, Berlin, FRG

The scrap metal site in the Charlottenberg area of Berlin (9) is contaminated with cyanide, arsenic, cadmium, mercury, PCB's, lead, volatile chlorinated hydrocarbons, and oil. Remedial operations have been active since December 1986, and consist of high pressure soils washing. A Netherlands company, Klockner Oecotec GMHB, has used this high pressure soils washing process to clean 100,000 tons of contaminated soils and removed over 95 percent of the pollutants. High pressure water jets are used to physically strip off hydrocarbons and heavy metal residues from the soils. This process is particularly promising for sandy/gravelly soils but is less effective for clays.

3.4 Unna-Boenen, North Rhine-Westphalia, FRG

The coke oven site at Unna-Boenen, North Rhine-Westphalia, is contaminated with aromatic hydrocarbons, tars, and acid resins (10). Volatile organic compounds comprise 5 percent of the waste by weight. The site is estimated to be 230,000 square meters.

A variety of technologies will be applied to the site, individually focused on the soil and aqueous phases of the contamination. A 50-ton per hour, transportable thermal unit will be used to process the soil and redeposit it on-site. The unit has been successfully pilot tested on similarly contaminated soils at a 7-ton per hour rate with a destruction effectiveness of approximately 98 percent. The soil is fed to a rotary kiln after mechanical pretreatment. The volatile matter is released from the soil at temperatures up to 600°C and released gases burnt at up to 1,300°C. The plant is scheduled to begin operation early in 1988. In addition, experiments to evaluate enhanced indigenous microbial degradation have been carried out. Although not completed at the time of writing, the initial results were not promising because of the difficulties in achieving contaminant/microbe contact. For contaminated water, a combination of air stripping, flocculation, sedimentation, and filtration are under consideration.

3.5 Pintsch-uil Site, West Berlin, FRG

The Pintsch-uil site (11) is located in an industrial estate of West Berlin. The recovery of used oils has been carried out since 1924. Residual products were left in several pits and subsequently seeped into the soil and groundwater. After an initial inspection in 1976, the water authorities in charge directed that groundwater samples be taken. Considerable groundwater pollution was detected and resulted in a requirement for groundwater cleanup. After thorough investigation of the site, three extraction wells with a maximum capacity of 40 m^3/h at 25 m WC were constructed.

A preliminary pollutant analyses of groundwater samples indicated extensive contamination with a wide variety of organic compounds. Results showed concentration for:

hydrocarbons	up to 16,000 mg/l
oils	up to 1,000 mg/l
phenols	up to 225 mg/l

In addition to the dissolved and undissolved oils and their compounds, groundwater contamination is primarily caused by volatile organic compounds (VOC's) and chlorinated hydrocarbons.

A pilot-plant with a capacity of 40 m^3/h was designed and constructed to purify ground and seeping water. It comprised an oil separator, flotation, countercurrent desorption with exhaust purification, and active carbon purification.

The other major part of this remedial activity comprised a full-scale cleaning of contaminated soils using a high-energy, water-based stripping process.

3.6 Province of Zurd, Rotterdam, The Netherlands

The contaminated soil at the former gasworks site in the Province of Zurd, Rotterdam, has been excavated and transported to the thermal destruction installation of Ecotechniek (12). The soil is contaminated primarily with polynuclear aromatics (PNA's) and complexed cyanides (ferri-ferro cyanides) caused by spillages and dumping of waste materials. The installation consists primarily of an internally heated rotating kiln (direct heat transfer) and an afterburner for the off gases; these operate at a maximum temperature level of 550°C and 1,100°C, respectively.

The main purposes of the demonstration project are:

a. To evaluate the treatment results and environmental aspects of the thermal cleaning of soil from a former gasworks site;

b. To develop a standard evaluation methodology for soil treatment plants in general. The evaluation was carried out using the methodology developed by TNO (13). The evaluation was carried out over several 2-day periods to include:

    (1) "easy to clean" sandy soil chiefly contaminated with polynuclear aromatic hydrocarbons (PAH's) from a gasworks site, and

    (2) "difficult to clean" clayey soil containing rubble and contaminated with PAH's and mineral oil.

A carefully designed statistical sampling program (12) was applied to both input and output soils. Gaseous emissions from the plant were also monitored. While removal rates above 95 percent were achieved for both soils, the "difficult soil" did not fully meet the statistically based requirement in regard to all parameters. However, it was suggested (13) that some modification of the "standard" sampling and evaluation method might be necessary. All air emission standards were met.

3.7 Asten Site, The Netherlands

The Asten site, in the province of Noord Brabant, is a petrol/(gasoline) station where the soil is contaminated with petrol containing small amounts of lead and a small quantity of diesel oil (14). The contamination was caused by a leaking tank. At least 30,000 liters of normal gasoline have leaked and about 1,500 cubic meters of soil have been contaminated. A 150-mm layer of free product was found in wells; 20,000 liters have been recovered by pumping. The soil is generally sandy and the water table is about 2.5 m below the surface. Concentrations in the soil vary mainly between 1,000 and 5,000 mg/kg of gasoline.

This project is aimed at optimizing the treatment of deeper layers of contaminated soil by enhancing in-situ microbial activity. The development of in-situ techniques is important for the reclamation of other similar sites. It is expected that the use of this technique will result in cost reduction since it will avoid costs for excavation and allow the petrol station to continue its operation.

A series of laboratory experiments has been carried out to determine the optimum treatment conditions for the soil, both in terms of microbial degradation, and infiltration of liquid/leaching of the soil, including tests using 200-mm diameter undisturbed soil columns taken from the site. A design for the cleanup operation has been produced based on these laboratory studies.

3.8 Wijster, The Netherlands

The Wijster site, in the province of Drenthe, was formerly occupied by a waste disposal company. The primary emphasis in the project is an improvement of landfarming methods for the biodestruction of gas oil, crude oil, and halogenated hydrocarbons (for example, hexachlorocyclohexane).

Contaminated soil will be excavated and spread over a drained sand bed. The sand bed will be isolated from the subsoils by a plastic membrane. Nutrients and the contaminated soils will be added, and the contaminated soils will be covered by an oxygen permeable plastic membrane or plants to prevent erosion.

3.9 Peak Oil, Tampa, Florida, USA

The Peak Oil site was previously operated by an oil refiner (15). Investigations started in 1984 showed that a waste lagoon contaminated with oil refining waste containing PCB's and lead (and also acetic acid) was contaminating the groundwater. The soil in the area is sandy and the groundwater table is very near the surface. The lagoon was drained of water and the contents were mixed with sand, soil, and lime to further adsorb water and to neutralize the acidic waste. The PCB's and lead contents of the treated waste were determined as less than 50 mg/kg and 10-15 mg/kg, respectively. Thermal destruction technology was selected for the cleanup of the waste lagoon.

The material then moves through the furnace where it is exposed to infrared radiation in multiple temperature-controlled zones. Zone temperatures are controlled by varying the input power (electric) to maintain preset zone setpoint temperatures. As the material moves through the furnace on the belt, it is exposed to the thermal environment necessary for oxidation of volatiles and solid organics.

The field phase of this project has been completely successful and a report is being prepared (December 1987) by the U.S. Environmental Protection Agency. The trial was successful although some difficulties were encountered with the exhaust gas treatment system.

As noted above, the lagoon material contained lead so that the treated soil/ash produced by the plant remains contaminated with lead. A final disposal route for the material has yet to be selected.

3.10 Eglin Air Force Base, USA

This jet fuel spill site is near the city of Fort Walton Beach in Florida (16). In 1984, a large area of dead grass appeared in the base petroleum storage area. The smell of fuel indicated that an underground fuel leak had occurred. Pressure testing of underground fuel lines revealed the source of the spill--a 1-liter per minute leak in a 150-mm diameter pipeline. The duration of the leak is unknown; however, soil and groundwater sampling indicates that as much as 100,000 liters of jet fuel have contaminated 6,000 - 8,000 cubic meters of soil and shallow groundwater.

The goals of this site demonstration (16) are to generate new data on the in-situ degradation of specific jet fuel components, to optimize nutrient/oxygen additions, and to maximize contact between nutrients, soil microbes, and fuel-soaked soils. Varieties of infiltration methods are being evaluated. The study will be conducted over a 2-year period to provide information on the lower limits of biological removal of fuel compounds. A control area has been established in a fuel contaminated area on the upgradient end of the site. The purpose of this control area will be to compare natural levels of degradation to the enhancement provided by nutrient/oxygen additions.

3.11 Verona Well Field Site, USA

The Verona Well Field site consists of several distinct contaminated areas within about 40 hectares (17). The well field itself contains 30 production wells that supply the entire city of Battle Creek, Michigan, including several major businesses. The site also includes a railroad marshalling yard and two solvent facilities. The Thomas Solvent Raymond Road (TSRR) facility is a former solvent repackaging and distribution facility. Solvents were stored in 21 underground storage tanks which were later discovered to be leaking. The TSRR facility is located about 1 mile upgradient of the well field. It is primarily a residential area surrounded by a few businesses.

Chlorinated hydrocarbons are the most significant environmental contaminants. Groundwater and soil contamination within the TSRR facility was found with VOC's as high as 100 mg/l and 1,000 mg/kg, respectively. The total estimated mass of organics in groundwater and soil at TSRR was 200 kg and 770 kg, respectively. The total contaminated area is about 900 m^2. The remedial action selected for the site comprises a groundwater extraction system (GWE) in conjunction with a soil vapor extraction (SVE) system. It was estimated that groundwater contamination could be reduced in 100 ppb in 3 years and the total contaminant mass reduced by 98 percent in 1-1/2 years.

The key components of the SVE system are extraction wells screened primarily in the unsaturated zone and a vapor phase activated carbon treatment vessel. A performance goal of 10 mg/kg total VOC's in the unsaturated zone was established. It is expected that this will be achieved in about 2 years.

Initial results indicate well influence of about 16 m radius, that recovery rates may be higher than estimated, but that the total amount of contaminants probably exceeds the estimate of 770 kg. A statistically based program of soil sampling has been established to assess overall performance and the progress of the removal operation.

3.12 Asahi Electrochemical Company Site, Japan

This site, located in Arakawa-ku, Tokyo (18), was used from 1917 primarily for the production of sodium hydroxide and bleaching powder, but also to produce soap, margarine, hydrogen chloride, and chlorinated organic compounds. From 1955 to 1966, the company operated the mercury electrolysis method to produce high quality sodium hydroxide.

The plant was closed in 1979 as part of a plan to relocate industry away from residential areas. The site was scheduled for reuse for a college, public park, municipal wastewater treatment plant, and community complex. During factory demolition in 1978, soils containing up to 15.6 percent (3.68 percent mean) mercury were found beneath the electrolysis vessel. About 840 m^3 of these heavily contaminated soils were drummed and taken to an off-site mercury roasting plant about 1,000 km away to recover mercury in 1979-80. About 54,000 m^3 of soils contaminated with mercury above 2 mg/kg were contained in an on-site pit after immobilization of soils with more than 10 mg/kg mercury (32,000 m^3) with sodium sulfide. The sides of the pit were lined with a concrete barrier wall. The base was formed by low-permeability natural strata. The pit was capped with EPDM sheets 1.5 mm thick and covered with 1.2 m of soil. Methane generation by organic matter in the buried soil required installation of gas extraction and monitoring wells.

4. EMERGING THEMES

A number of the projects involve microbial treatment of groundwater or soil. In some cases, the intention is to achieve treatment in-situ; in others, the treatment is achieved after extraction/excavation. It was generally agreed during the discussions at the Washington, D.C. meeting, that microbial treatment of many contaminants is a proven possibility under the right conditions, but that the major limiting factor to practical application is the difficulty of ensuring contact between bacteria and contaminants, both in terms of getting the treatment agents to penetrate the soil and of getting the contaminants into solution, and thus available to the bacteria. The practical tests of application methods (Elgin Air Force Base) and thorough laboratory trials prior to field application (Asten petrol station) are thus important complements to the field demonstrations of microbial treatment, as are the basic studies and the concentrations to be dealt with.

5. CONCLUSIONS

The Pilot Study is working well and there has been a fruitful exchange of information and views between the participants. This exchange is based on the active cooperation and participation of government officials, academics, industrialists, and researchers drawn from the participating countries. It is hoped that further countries will recognize the benefit of such international cooperation and associate themselves with the study, either as full participants contributing demonstration projects themselves, or in an observer role. Inquiries about the study should be addressed to the Pilot Study Director, Donald E. Sanning, or to the national contacts listed in Table 3.

6. ACKNOWLEDGMENTS

Space does not permit the listing of all those contributing to the Pilot Study, but the Pilot Study Director would like to give particular thanks to the representatives of the co-pilot countries--K. Stief of the FRG, Ms. E. Soczo of The Netherlands, Ms. Elfreda Naberhaus for typing the paper, and to those reporting on the individual projects. (See References.)

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Country	Project	Technologies, etc.
Canada	Ville Mercier	Groundwater treatment
Denmark	Skrydstrup	Groundwater and soils cleanup, microbial techniques
France*	Lot River Viviez, Aveyron	Pump and treat, recovery/recycling; Stabilization/solidification evaluation at four sites in France
FRG	Charlottenberg, Berlin	Soils cleanup (high pressure water washing)
	Unna-Boenen coke oven plant	Soils cleanup by thermal treatment, microbial treatments
	Pintsch-uil, West Berlin	Groundwater treatment, soils cleanup by high-energy water stripping process
The Netherlands	Rotterdam	Thermal treatment of soils
	Asten	In-situ and post extraction cleanup of groundwater (and associated soils) by microbial techniques
	Wijster	Microbial land farming
USA	Peak Oil	Thermal treatment of soil
	Eglin Air Force Base	Groundwater/soils cleanup by in-situ microbial treatment
	Verona Well Field	Soils/groundwater cleanup by soil vapor extraction
Japan	Asahi Electrochemical Company	Soils treatment by thermal and chemical means

*Not added until March 1988.

Table 1 Original (November 1985) NATO/CCMS pilot study demonstration projects by country.

Type of Treatment	Former Use/Problem	Location	Contaminant(s)
France			
Pump and Treatment --Recovery/Recycling (groundwater)	Zinc smelting plant	Lot River Viviez, Aveyron France	Zinc, cadmium
Stabilization/Solidification --Solidification evaluation	Petroleum refinery wastes/agricultural chemical manufacturing/ tannery	4 sites, France	Acid, tars, heavy metals
Federal Republic of Germany			
Soil Treatment by Extraction --High-pressure soil washing	Former disinfectant manufacturer	Goldbeck-Haus Hamburg - Winterhude Federal Republic of Germany	Phenols
The Netherlands			
Microbial Degradation --Biological pretreatment of groundwater	Lindane manufacturer	Bunschoten, The Netherlands	Lindane, benzene
Soil Treatment by Extraction --In-situ extraction	Photographic paper manufacturer	Soestduinen, The Netherlands	Cadmium
United States			
Chemical Treatment --K-PEG Technology (in soils and liquids)	Chemical recycling facility and road oiling	Gary, Indiana (Conservation Chemical) and Wide Beach, New York, United States	PCB's, chlorinated hydrocarbons
Volatilization --Radio frequency volatilization	Abandoned firefighter training area	Volk Air Field Wisconsin, United States	Gasoline/ kerosene mixture
Stabilization/Solidification --Solidification (Hazcon process)	Lagoons, landfarm, and spill areas	Douglasville, Pennsylvania, United States	Lead, oil, volatile organics, PCB's

Table 2 New (March 1988) NATO/CCMS pilot study demonstration projects by country.

Canada*	Mr. J. Schmidt	Environment Canada, Burlington, Ontario
Denmark*	Dr. S. Vedby	National Agency of Environmental Protection, Copenhagen
FRG*	Mr. K. Stief	Unweltbundesamt, Berlin
France*	Mr. R. Goubier	ANRED-France, Angers
Greece	No official contact	
Italy	Dr. Giuseppe Giuliano	The Institute for Research on Water, Rome, Italy
Japan*	Mr. T. Ikeguchi	Institute of Public Health, Tokyo
Netherlands*	Ms. E. Soczo	RIVM, Bilthoven
Norway	Mr. M. Helle	State Pollution Control Authorito, Oslo
Spain	No official contact	
United Kingdom	Mr. J. Mather**	British Geological Survey, Wallingford
USA*	Mr. D. E. Sanning	U.S. Environmental Protection Agency, Cincinnati, Ohio

* Active participant
**Contact for information, not official government representative

Table 3 National Contacts

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8. Christiansenk (?) and Vedby, S. "Skrydstrup Chemical Waste Disposal Site," in Ref. 5.

9. Heimard, H.J. "High Pressure Soil Washing Process for Cleaning Polluted Soil in Berlin," in Ref. 5.

10. Ronge, J. "Strategy for Remedial Actions at Contaminated Abandoned Industrial Sites," in Ref. 5.

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14. Verheul, J. and Eikelboom, D. "In-situ Biorestoration of a Subsoil Contaminated with Oil," in Ref. 5.

15. James, S.C. "Demonstration of Electric Infrared Incineration at Peak Oil Superfund Site," in Ref. 5.

16. Downey, M. "Enhanced Biodegradation of Jet Fuels - Eglin Air Force Base, U.S.A.," in Ref. 5.

17. Tanaka, J.C. "Verona Well Field Superfund Site: Soil Vapor Extraction System," in Ref. 5.

18. Ikeguchi, T. "Former Electrochemical Industry Site, Arakawa-ku, Tokyo," in Ref. 5.

INFLUENCE OF MACRO-PORES ON WATER BALANCE AT DISPOSAL SITES

Manoj Mishra[1] and Carol J. Miller[2]

ABSTRACT

An hydraulic description of moisture transport through the compacted clay layers of a landfill cover liner is presented. Previous descriptions have assumed that micro-pore processes control the transport. The present description incorporates the influence of macro-pores, as created by desiccation cracking. The existence and importance of cracks is documented using previously collected laboratory and field data. Although micro-pore flow can be modelled confidently using Darcy's Law, its application to macro-pore flow has not been documented. The variety of modes and mechanisms of flow that are operative in flow through cracked clay liners is discussed, and analogies are made to familiar concepts in open channel flow theory. The discussion provides the conceptual framework for the development of a numerical model to simulate both the micro- and macro-pore phenomena.

INTRODUCTION

The groundwater contamination potential of a landfill depends on many parameters. Two most important factors are the leachate migration out of the landfill and the corresponding contaminant concentration in the leachate. The former factor is controlled by the design, construction, and maintenance of the landfill, while the latter depends primarily on the waste characteristics and available moisture. This paper focuses on leachate migration out of the landfill into the native soil, since it is under more direct control of the design engineer.

The bottom liner provides the final barrier to leachate migration to the native soils and groundwater. The bottom liner consists either of a layer of compacted clay alone or its combination with a synthetic membrane. Hazardous waste landfills are required to have a double synthetic membrane. Both synthetic and natural materials may degrade while functioning as a bottom liner. Gray (1984) has reported that leachate can increase the permeability of compacted clay liners. The presence of hydraulic volume defects, such as desiccation and syneresis cracks, can also increase the leakage through a liner. He

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reports commercial polymeric membranes to be adversely affected by the wastes that are highly ionic, contain salts, strong acids/bases, or organic compounds. Oily wastes, in particular, tend to cause swelling and degrade desirable physical properties (e.g., elongation and resiliency) in the membrane liners. On the basis of the work of Haxo (1980, 1981), Gray (1984) has concluded that no single available lining material appears suitable for the long term impoundment of all wastes. With such uncertainties surrounding the performance of a landfill's bottom liner, the only reliable measure of the potential damage due to leachate migration is the volume of leachate generated in the landfill.

Leachate is generated with the moisture available from two sources: liquid in the disposed waste and infiltration through the top of the landfill. From the initial construction of a landfill until a short period after its closure, both of these sources contribute to the leachate generation. However, once gravity flow has removed most of the initial moisture content of the waste, water infiltrating from the top of the landfill becomes the only source of moisture responsible for leachate generation. The initial moisture content of the waste can be determined while dumping the waste. Hence, the volume of water infiltrating a landfill remains the only unknown to be determined in estimating the volume of leachate generated.

Infiltration from the top of landfill is controlled by the cover liner. The cover liner consists of a layer of the compacted clay alone or its combination with a synthetic membrane. A synthetic membrane is included for all hazardous waste facilities, and acts as a "back up" to the clay. A sand layer is included to facilitate lateral flow of the leakage towards collection pipes which route the moisture off-site.

The synthetic membrane of the cover liner can effectively perform its function only as long as it remains intact. Haxo (1981) has reported on the puncturing of synthetic membranes due to the compactive efforts of construction equipment on the overlying clay layers. Gray (1984) has also reported on the increased leakage through these membranes due to seams and pin holes. Infiltration may also pass through faulty seals around the manholes, vent pipes, and other items that pass thorugh the cover liner. These findings considerably reduce the reliability of the synthetic membranes as a barrier for moisture movement. Hence, the ultimate responsibility of reducing the generation and the leakage of the leachate from a landfill shifts to the clay cover liner. This conclusion makes the estimation of the leakage through the clay cover liners crucial for assessing the generation of leachate and, hence, the maximum possible leakage from a landfill.

CONSIDERATION OF PREVIOUS WORK

Mathematical Modelling of Leakage Through the Cover Liner

Mathematical modelling of the leakage through the clay cover liner has been attempted by Remson et al. (1968), Fenn et al. (1975), Perrier and Gibson (1980), Skaggs (1980), Miller (1984) and Schroeder (1984a and b). The first three of these researchers have used the water balance method, without considering the mechanism of the fluid flow through the porous media, to estimate the leakage. This approach has been found to result in gross underestimates of the field conditions (Gee, 1981; Gibson and Malone, 1982). Skaggs (1980) improved over the earlier attempts by incorporating Darcy's equation in his model to compute the rate of moisture movement through the saturated porous media. Both Miller (1984) and Schroeder (1984a and b) used a quasi-two-dimensional, deterministic approach to model flow through the cover liner. Schroeder's model assumes the clay liner to be saturated, while Miller's model covers the full range of saturation. However, Schroeder models the surface hydrology, while Miller's model requires the user to provide the surface inputs. In Miller's model it is assumed that the surface inputs (i.e., the flux boundary condition) have been solved for apriori, using a model of the surface hydrology.

All of the above-mentioned researchers have assumed the clay liner to be a homogeneous mass of clay with uniform hydraulic properties. The hydraulic conductivity values used in the models are determined in the laboratory and, hence, do not represent the actual field values. In a recent study, Daniel (1984) found that fluid moves much faster through the clay liners than predicted with the laboratory determined values of hydraulic conductivity. He discovered that the actual hydraulic conductivities of the clay liners, back calculated from the field leakage rates, were from 5 to 10,000 times larger than those obtained from the laboratory tests on either undisturbed or recompacted samples of the clay liners. Researchers attribute this increased hydraulic conductivity in the field to the presence of the macro-pores in the clay mass (Daniel, 1984; Boynton and Daniel, 1985).

A macro-pore can be defined, typically, as a void space in the soil mass which is larger than that formed by the individual soil particles placed side by side in a soil matrix. Cracks, fissures, root holes, and wormholes are some examples of the macro-pores generally found in a soil mass.

Flow Through Soil Macro-pores: A Recognized Phenomenon

Rapid movement of surface-applied water through the macro-pores may be a novel speculation for investigators estimating leakage through landfill liners. However, it is a long established fact for agricultural and petroleum engineers. According to Beven and Germann (1982), the concept of rapid movement of water through macro-pores was first described by Schumacher (1864) who wrote: "...the permeability of a soil during infiltration is mainly controlled by big pores, in which the water is not held under the capillary forces." Later, Lawes et al. (1882), Horton (1942) and Hursh (1944) also reported on the rapid movement of surface applied water through macro-pores. Hursh (1944) has clearly stated that: "...in considering flow through upper soil horizons, the formulas of soil mechanics do not generally apply. Here porosity is not a factor of the individual soil particle size but rather of structure determined by soil aggregates which form a three-dimensional lattice pattern. This structure is permeated throughout by biological channels which in themselves also function as hydraulic pathways. A single dead-root channel, wormhole or insect burrow may govern both the draining of water and escape of air through a considerable block of soil."

Recently, Edwards et al. (1979) have reported that Gardner (1962) also showed that the holes and the cracks, that are open at the surface, can definitely move significant volumes of the free surface water much deeper and faster than can be calculated by the Darcy-type solutions alone. The tracer studies of Ritchie et al. (1972), Blake et al. (1973), and Quisenberry and Phillips (1976) also revealed that the surface-applied water flows through the interconnected macro-pores. Miller (1986) has also mentioned that if flow through the cracks in a landfill liner is significant, her numerical model is expeced to underpredict the leakage through the clay liners.

The first reported observation of flow through macro- pores (Shumacher, 1864) is contemporary to the initial discussion of flow through the micro-pores in a porous medium (Darcy, 1856). Although the importance of micro-pores in fluid flow through porous media is now well understood and quantified through Darcy's equation, not much has been achieved in understanding and quantifying the flow through the <u>macro-pores</u>. Some efforts made by various researchers to quantify the influence of the macro-pores on infiltration through a soil mass are presented subsequently.

Flow through the macro-pores, until now, has been of major interest only to agricultural and petroleum engineers. Differences in the nature of the porous media encountered by the investigators in these two fields have resulted in two distinctly different approaches to model the flow through the porous media containing the macro-pores.

Petroleum engineers encounter fluid flow through fractured rocks under considerable hydraulic gradients. Since the hydraulic conductivity of the rock matrix is negligible, rocks have been treated as impermeable. The fractured rocks are modelled as a network of intersecting horizontal and vertical fractures (Wilson and Witherspoon, 1974) and the individual fracture as a set of parallel plates (Huitt, 1956). Therefore, the investigations by the petroleum engineers concentrate mainly on the effect of the surface roughness (Huitt, 1956); the variations in the aperture size and the shape (Neuzil and Tracy, 1981; Long et al., 1985), and their distribution (Sagar and Runchal, 1982) on the fluid flow through the fractured rocks.

Agricultural engineers are concerned about the influence of root holes, wormholes, and surface connected cracks on the infiltration of surface-applied water into the soil peds in the root-zone of the agricultural soils. Therefore, their investigations concentrate more on the absorption of the water from the water carrying macro-pores into the surrounding soil matrix than on the flow through the macro-pores themselves. The flow conditions encountered in a clay cover liner are more similar to that in an agricultural soil than in a fractured rock. Therefore, only the findings of the agricultural engineers and their possible applications to the clay cover liners are discussed here.

Davidson (1984) has presented a model to trace the motion of a wetting front travelling into a soil containing regularly spaced, water filled cracks of negligible width. At the beginning of the wetting process, two sharp fronts, separating the wetted region from the unwetted one and advancing in intersecting directions are presumed to originate and grow away from the top surface and the crack. Initially, the former advances as a one-dimensional flow and the latter exhibits a mushrooming effect with the lateral advance near the tip of the crack being greater than at the base.

Since the macro-pores considered in the Davidson's model are isolated, of negligibly small aperture and closed bottom, the question of the drainage <u>through</u> them is meaningless. Hence, Davidson's findings are applicable to the clay liner problem only for understanding the movement of the wetting front in individual soil blocks surrounded by the cracks.

An attempt to mathematically model the influence of the macro-pores of finite aperture on the soil-water movement was made by Edwards et al. (1979). Using a level field with a large number of regularly spaced, round, vertical wormholes, they considered a single hole and a small cylindrical column of the soil around it. Their model simulates the flow of water, falling on the top surface of this vertical soil column, by using Richard's equation with many assumptions: the soil properties and the initial water content are uniform with the depth, the soil remains unsaturated throughout the rainfall, and the soil column receives heavy precipitation of constant intensity.

This approach was improved by Beven and Clarke (1986). In addition to the infiltration from the sides of the circular macro-pore, vertical infiltration from its bottom is also considered. They modeled the infiltration into a soil profile due to the stochastic variance in spatial occurance of the macro-pores. The results obtained by considering a single circular channel are generalized by considering the statistical population of channels with a given probability distribution of radii and depth. The vertical axes of the cylindrical channels are assumed to intersect the horizontal soil surface at the points having poisson distribution. The assumptions made by Beven and Clarke for the model formulation restrict the model applications to a soil containing vertical, isolated, and circular conduits. In addition, since the model considers the infiltration into the soil matrix from the macro-pores and not the flow <u>through</u> them, its direct application to the liner problem is not possible.

Bouma and Dekker (1978) had observed that the depth and the patterns of the vertical penetration of water along the cracks in the dry, heavy clay soils were a function of the intensity and the duration of the rainfall event. They sprayed different quantities of dilute Methylene-blue solution, using different intensities, over 37 rectangular plots of 100 cm by 50 cm. On excavating these plots to a depth of 1 m, they observed that, in a crack, instead of flowing along whole of the crack length, the solution was visible only along a few 5 to 7 mm wide bands. The area of these bands, made visible by the blue stains, was added together to get the total area, S, through which the transfer of water from cracks to the surrounding soil matrix took place.

Hoogmoed and Bouma (1980) utilized the findings of Bouma and Dekker (1978) to estimate the transfer of surface-applied water to a soil profile containing surface-connected cracks. In their model, they incorporated the models of Van Keulen and Van Beek (1971), to estimate the vertical flow through soil layers, and Van der Ploeg and Benecke (1974), to estimate the horizontal absorption of water by an initially dry soil. They used the technique of Bouma and Dekker (1978), described above, to determine the wetted surface area, S, of water carrying cracks through which the absorption into sorrounding soil matrix takes place. Their model assumes that:

1. the soil is initially dry,
2. the ponding of the water starts when the rainfall rate exceeds the rate of infiltration,
3. the water starts flowing in the cracks when the depth of ponding reaches a preset, threshold value, of 0.2 cm based on visual observations, Hoogmoed and Bouma (1980)
4. all of the excess water flows into the cracks, and
5. the drainage from the soil starts when the flow rate into the cracks exceeds the total horizontal absorption of all layers.

The model simulates the movement of the surface applied rain-water through a 20 cm thick layer of soil containing cracks. For calculation purposes, this soil layer is divided, from bottom to top, into 11 layers: 9 of 2 cm thickness and one each of 1.5 cm and 0.5 cm. Water reaching the bottom of the soil layer is considered as the drainage water.

The results of this simulation show that depending on the initial moisture content of the soil and the rainfall intensity, the rate of drainage can be up to 97% of the rainfall intensity. High drainage rates like this, although in agreement with the corresponding drainage rate (98%) measured by Bouma et al. (1978), are not expected from a landfill liner because their assumption of all of the excess water flowing into the cracks is not valid there. Field observations by the authors indicate that, mainly because of the liner geometry, a part of the excess water always flows off the landfill surface.

Another outcome of this model, the one we may find very useful in modelling the leakage through the cracked liners, is that the vertical infiltration is more important than the horizontal absorption for heavy clay soils. With the passage of time, as the wetting front travelling down through a soil mass reaches a particular depth, the horizontal absorption at this depth reduces. Hoogmoed and Bouma (1980) expect that for the soils with a texture coarser than that of the clay soils considered in their model, the horizontal absorption may be quite significant. This correlation of the soil texture with the horizontal absorption can form the basis for the assumption that the horizontal absorption in the liner clays, having a very fine texture and low permeability, will be negligible. This assumption will help reduce the complexity of the mathematical model for estimating the leakage through the cracked liners without adversely affecting the leakage estimates.

The model of Hoogmoed and Bouma (1980) has limited utility for estimating leakage through cracked liners. Information about the depth and the amount of vertical infiltration into the surface peds can be obtained from it, but an accurate estimate of flow into the cracks is not possible.

Other authors, such as Gray (1984), have suggested the concept of "equivalent hydraulic conductivity" to account for the complex interaction of micro- and macro-pore flow in a cracked liner. Such an equivalent value would require a laboratory determination whereby the total flow exiting the cracked system would be measured. The equivalent hydraulic conductivity resulting from such a determination would be the coefficient in Darcy's equation resulting in an equality between flow and hydraulic gradient, and would, in some sense, reflect an "averaging" of the two flow mechanisms. The problem is that the equivalent hydraulic conductivity would apply only to those specific laboratory conditions for which it has been calibrated. For different rainfall intensities, micro-pore structures, clay types and crack geometries, the relative importance of macro-pore vs micro-pore flow will change. Because the concept of equivalent hydraulic conductivity does not incorporate the actual physical mechanisms underlying the flow processes, it is not generally applicable.

Bronswijk (1988) investigated the agricultural implications of moisture transport in soil macro-pores. Bronswijk uses a water balance approach to model the vertical movement of water through a clay soil, from the surface to the underlying groundwater. The primary contribution of his work is the introduction of the concept of a "shrinkage characteristic". He demonstrates that this characteristic curve may be used to relate moisture fluctuations in the soil to volume changes, which comprise both subsidence and cracking. Using assumptions of crack depth, the aggregated surface area of cracks is determined. The rainfall is then dynamically partitioned between the cracks and soil matrix. Bronswijk neglects any horizontal absorption as the water flows down the cracks, assuming it all accumulates at the bottom of the crack. Also, this approach assumes negligible resistance to flow through the cracks and that flow into the cracks is limited only by the surface area of the cracks. Surface runoff is neglected, as Bronswijk assumes a horizontal soil surface. However, landfill cover designs require a surface slope to promote runoff.

As revealed by this literature review, existing models of flow through macro-pores are not sufficiently developed and suitable for the simulation of the leakage through clay liners. Either their scope is limited to estimating the infiltration into the soil matrix, from the macro-pores, or their methodologies limit their utility to the liner problem. Therefore, the mathematical modelling of the leakage through the clay liners demands a fresh and innovative investigation of flow through macro-pores specific to the liner environment. Such an investigation requires an understanding of the physical structure of the macro-pores present in the clay liner.

DESCRIPTION OF THE PHYSICAL SYSTEM

A clay liner consists of one or more layers of clay characterized by high plasticity index and compacted to yield very low hydraulic conductivity. Multi-layer liners are constructed by the sequential compaction of the individual clay layers on top of one another. In the case of landfill cover liners, the topmost layer of the compacted clay is overlain by organic soil, also known as the topsoil, to support the vegetative cover on the landfill. The mineralogical composition, the degree of compaction, and the physical and biological environment of the liner clay and the topsoil differ considerably. As a result, the macro-pores in these two components of the cover liner also differ in their origin and type. The mechanisms of flow through them are also different and a clear understanding of these different flow mechanisms is vital for the mathematical modelling of the leakage through the clay liners.

The topsoil layer of a cover liner is densely populated by the plant roots and provides habitat to the worms and the burrowing animals associated with the vegetation. This situation leads to the formation of the macro-pores that can be approximated by circular conduits. A significant fraction of the runoff generated on the landfill surface may be directly conducted to the topmost layer of the compacted clay by a network of such interconnected conduits (Fig. 1). In due course of time, a part of this infiltrated water may also reach the top of the compacted clay layer by seeping out from the soil matrix into the interconnected channel network (Fig. 2). Thus, the total amount of leakage through the topsoil layer is expected to be much more than estimated by the existing models that do not consider the flow through the macro-pores. This unaccounted increase in leakage through the topsoil layer also increases the importance of the underlying compacted clay layers for impeding the further movement of the moisture. Therefore, a careful investigation of the macro-pores in the compacted clay layers, and the mechanisms of flow through them, becomes even more important.

The compacted clay portion of a landfill liner consists of one or more layers of clay compacted sequentially on top of one another. The density and the moisture content of each of these layers is kept close to their respective optimum values to obtain low hydraulic conductivity. The compaction of a layer takes several days or longer and may not be immediately followed by the compaction of the next layer. During all this time, the compacted clay layer is exposed to the atmosphere. Consequently, it experiences the drying associated with wind exposure and temperature effects and subsequently develops desiccation cracks (Photo 1).

During their field studies (Miller, et al., 1987), the authors have found that the desiccation cracks - uniformly distributed over the entire compacted clay layer - can exceed one centimeter in width (Photo 2). These cracks form vertical channels which may penetrate the entire depth of a compacted layer. Depending on the duration of the atmospheric exposure and the local climate, the development of desiccation cracks can finally break the initially homogeneous compacted clay layer into much smaller individual clay peds. These peds are completely isolated from their surroundings by means of the cracks and can virtually be removed from the clay layer with little effort and negligible interaction with the surrounding soil (Photo 3). In such a physical state, the clay layer contains vertical channels distributed throughout. Such channels can significantly reduce the effective thickness of the compacted clay liner. In addition, they can quickly conduct a significant portion of the surface runoff directly to the bottom of the clay liner.

Shrinkage of the compacted clay during the desiccation process may actually reduce the hydraulic conductivity of the clay forming the individual peds. However, the channels formed around the peds, by the desiccation cracks, increase the hydraulic conductivity of the compacted clay layer as a whole. As a result, during a rainfall event, the infiltration through the matrix of the individual peds may be desirably low while the volume and the rate of leakage through the whole layer may be unacceptable. *In a situation like this, the conventional* laboratory tests, on small samples from individual peds, may indicate compliance with the regulatory requirements for hydraulic conductivity while, in fact, that is not the case.

Some investigators have suggested that the problem of clay cracks is limited in liners because of the multi-layer approach. They reason that the multi-layers will buffer lower clay from environmental effects and the propagation of cracks. However, when a cracked clay layer is overlaid by a layer of the loose clay to construct the next compacted layer, not all of its cracks are completely filled up. In addition, the depth of the clay layer is greater than the length of the sheepsfoot in the rollers used for the compaction. Therefore, the channels in the underlying old clay layer stand a good chance of remaining mostly undestroyed and unsealed. They are merely filled up by the loose clay and remain highly permeable as compared to the clay matrix surrounding them. In due course of time, the new layer also develops desiccation cracks and the associated channel network. Now, the flow paths in the combination of the two layers appear similar to those in Figure 3.

During a rainfall event, the channels in the top layer deliver a portion of the surface runoff to the underlying, loose clay filled, channels. When the rainfall ceases, some of the runoff is stored in channels in the top layer and finally leaks through the underlying loose clay filled channels. The construction of each new layer adds an additional layer of loose clay filled channels to the existing flow paths. Each of such successive layers increases the delay in the onset of leakage due to a rainfall event. Successive reduction in the volume of leakage is also expected, and is brought to a minimum by the deposition of the topsoil layer. At this stage, the macro-pores constituting the flow paths responsible for leakage through the top-cover clay liner would look somewhat similar to those in Figure 4.

MODES AND MECHANISMS OF FLOW THROUGH LINER MACRO-PORES

Field investigations of the physical structure of the landfill cover liner, and the macro-pores in it, clearly indicate the existence of three distinctly different modes of flow through a liner. They are encountered during the different stages of the liner construction and, during each mode, a particular combination of six different flow mechanisms is responsible for the leakage through the compacted clay liner. Therefore, a clear understanding of all three modes and the six mechanisms is indispensable to the development of a mathematical model which accurately simulates the physical problem. All three modes and their associated flow mechanisms are briefly described below.

Flow through channels in the surface-exposed macro-pores of clay cracks comprises one mode of flow. It commences following construction of the lowest layer of compacted clay and persists at the top of each subsequent clay layer until the deposition of the topsoil layer. Therefore, it is operable only during the cover construction.

For a short initial period during a rainfall, the nature of flow in the channels is speculated as similar to that of free-falling water over a weir (Fig. 5). The duration of this period of flow is width-dependent and varies with the intensity and duration of a particular rainfall event. Since the rate of flow in interconnected channels is limited by the outflow at the intersections, a channel starts filling up when the rate of inflow into it exceeds the rate of outflow at its intersections. The narrower channels are filled up earlier than the wider ones and the free-falling flow is discontinued as soon as they are full. Consequently, the free-falling type of flow is most important in the wider channels.

If the velocity of the surface runoff provides sufficient kinetic energy, the second mechanism commences. With this flow mechanism, the volume of the runoff passing over a crack is divided into two parts. One part enters the crack as weir flow described previously, while the other part passes over the crack to continue as overland flow down the slope (Fig. 6). The value of the critical velocity at which the flow partitioning takes place varies with the crack width. The proportion of the runoff volume comprising each part depends on the flow velocity and the crack width.

After the macro-pores have filled with water (expected to be a very short time after the initiation of rainfall), the nature of flow in the associated channels is similar to that in closed rectangular conduits with permeable surfaces, flowing full. This flow mechanism requires sufficient ponding of the water, on the upper surface of the compacted clay layer exposed to the atmosphere, to provide, at least, a threshold hydraulic head (Fig. 7). The flow into the water filled openings is controlled primarily by the Darcy type flow into the underlying loose clay filled channels and horizontal absorption into the surrounding peds.

A second mode of flow through the macro-pores in a clay layer is operable in the loose clay-filled channels and is initiated due to the movement of a wetting front through the overlying clay layers. The channels in the overlying layers can be either loose clay filled type, those directly exposed to the atmosphere, or the circular conduit type macro-pores in the topsoil. Therefore, this second mode of flow is applicable only after the compaction of the second lowest layer, in a multi-layer liner, or after the deposition of the topsoil layer. The flow *mechanism is similar to that in the Darcy flow through the soil* micro-pores. Factors controlling the flow rate include: hydraulic head in the overlying clay layers, the average minimum width of the channels, and the hydraulic conductivity of the clay filler. It is assumed that the wetting front is advancing from above, and there is no restriction to flow at the bottom of the channel.

The third mode of flow exists only in the topsoil layer, where the macro-pores are primarily due to worm holes and root holes. The circular conduit type macro-pores do not develop until growth of the cover vegetation. Until then, infiltration is due to Darcy flow through the soil matrix. As time passes, the penetration of the plant roots and the activities of worms and burrowing animals provide the circular conduit type macro-pores in the topsoil. The flow mechanisms are identical to those described for the first mode, in channels directly exposed to the atmosphere. A portion of the water flowing in the circular conduit type macro-pores infiltrates into the surrounding soil matrix. *However, after the rainfall, the water from the soil* matrix infiltrates into the macro-pores resulting in an increase in the volume and the duration of the leakage from the topsoil layer.

CONCEPTUAL DEVELOPMENT OF A MATHEMATICAL MODEL

Any mathematical model used to simulate leakage through cover liners should incorporate both the micro- and the macro-pore flow mechanisms. The mathematics describing the flow through the clay micro-pores is well developed and some of the existing models (Miller, 1984; Schroeder, 1984; Korfiatis and Demetracopoulos, 1986) do use it. However, the development and the effective concatenation of mathematical models of flow through the variably sized pores, as present in the clay liners, is essential for the reliable prediction of leakage.

The volume of moisture passing through the topsoil layer *represents the maximum volume of leakage through the entire* compacted clay cover liner. The total annual volume of leakage in a particular year is the sum of the leakage due to each individual rainfall event occuring in that year. The volume of the leakage due to a particular rainfall event depends upon the duration of the rainfall and the rate of leakage through the topsoil layer. This rate of leakage through the topsoil layer, L_{ts}, can be written for a given time, t, as:

$$L_{ts} = L_{ma} + L_{mi} - S$$

Where:

L_{ma} = Rate of inflow into the <u>ma</u>cro-pores exposed at the surface of the topsoil layer.
L_{mi} = Rate of infiltration into the <u>mi</u>cro-pores in the topsoil layer.

S = Rate of change of the moisture content of the soil layer. S has a positive value when the moisture content is increasing. It is negative when the soil moisture content is decreasing and zero when steady state flow conditions prevail.

When the rainfall ceases, the water stored in the soil layers becomes the sole source of leakage from the soil layer. For this special case, the rate equation, given above, can be rewritten as:

$$L_{ts} = -S$$

Some of the previously discussed models of flow through macro-pores may be further developed to replace the symbolic rate expressions given above. Such a development of existing models for the leakage estimation calls for the determination of certain parameters, including:

1. the range of the macro-pore aperture sizes and their spatial distribution on the topsoil surface,
2. the range of the macro-pore depths and their spatial distribution into the topsoil layer,
3. the relative proportions and the distribution patterns of the isolated and the interconnected macro-pores, and
4. the density and the distribution of the intersections forming the interconnected network of the macro-pores.

However, the unpredictable development of the plant roots and the animal activity prohibits a reasonable estimation of these parameters. In addition to this, the interaction of the flow through the micro- and the macro-pores is a complex, time-dependent phenomenon. It is an important factor affecting the rate and duration of the leakage through the topsoil layer. However, it is difficult to model mathematically because of the macro-pore intersections and their unpredictable occurence. These factors, jointly, prohibit the development of a mathematical model giving reasonable estimates of the leakage through the topsoil layer of the cover liners.

Predictions of leakage through clay cover liners are hindered by lack of a quantitative model for topsoil leakage calculations. However, valuable information regarding the worst case condition of leakage in the clay liners can still be obtained. This rate of leakage, R_{cc}, is obtained in absence of the topsoil layer and can be sybmolically expressed as:

$$R_{cc} = R_{ma} + R_{mi} - S$$

Where:

R_{ma} = Rate of inflow into the <u>ma</u>cro-pores exposed at the top surface,

R_{mi} = Rate of infiltration into the <u>mi</u>cro-pores at the top surface of the compacted clay layer combination.

This equation exactly represents the physical situation for a landfill that is still in the process of cover construction, or for a landfill that has a badly cracked, eroded, and nonfunctional topsoil layer. For a completed landfill, this expression represents the upper limit of leakage expected through the compacted clay cover liner.

Mathematical modelling of leakage through a given combination of compacted clay layers requires the replacement of these symbolic expressions by their mathematical equivalents. This, in turn, necessitates the determination of parameters similar to those discussed in the case of the mathematical modelling of the leakage through the topsoil layer. However, the determination of the parameters for the macro-pores in the compacted clay layers, in contrast to that in the topsoil layer, is relatively simple. Therefore, after some reasonable assumptions, it is possible to develop an acceptable mathematical model for the leakage through the compacted clay layers.

The desiccation cracks, as opposed to the circular conduit type macro-pores in the topsoil layer, develop in a rather systematic manner. Their complete development leads to the formation of the peds (Photo 1 and Photo 3) that can be approximated by regular geometrical shapes. This approximation facilitates the determination of the density of thepeds on the surface of a clay layer. Since the water conducting channels are formed by the desiccation cracks around the peds, the determination of the ped density leads to the determination of the density of the vertical channels on the clay surface. The ped formation also stabilizes the width and the depth of the associated channels at their respective maximum values. Therefore, the regular geometry of the peds also facilitates the determination of the channel widths.

A major problem in the mathematical modelling of the leakage through the topsoil layer, as discussed earlier, was the modelling of the interaction of the flow through the micro- and the macro-pores. It was further complicated by the time dependency of the interaction process and the undeterminable occurance of the macro-pore intersections. However, in the case of the modelling of the leakage through the compacted clay layers, the hydraulic characteristics of the clay and the stepwise compaction procedure simplify this problem.

The simulation studies by Hoogmoed and Bouma (1980) have found the vertical infiltration in the heavy clay soils to be more important than the horizontal absorption from the macro-pores. They expected that for the soils with a coarser texture than the clay soils considered in their model, the horizontal absorption may be quite significant. On the basis of their correlation of the soil texture with the horizontal absorption, we can tentatively assume that the horizontal absorption in the liner clay, having a very fine texture and low permeability, will be negligible. This assumption simplifies the mathematics describing S.

The mathematical replacement for micro-pore flow term, R_{mi}, is well developed and readily available in the literature. However, the mathematics describing the flow through the macro-pores in the compacted clay layers is nonexistent. The authors are currently developing the mathematical models describing flow through both types of channels - those directly exposed to the atmosphere and vertical, loose clay filled channels. The development of these two models will be used to provide an "upper limit" estimate of leakage through the compacted clay component of a cover liner. This estimate provides a vital input to the design of the liner drainage and leachate collection systems. In addition, it provides important information regarding the maximum possible leakage through the bottom liner systems, and, hence, the assessment of the risk to the environment from a proposed or existing landfill.

CONCLUSIONS AND RECOMMENDATIONS

The importance of clay liners to the containment of landfill waste has been documented. Assessment of the containment efficiency of a proposed landfill requires the determination of the volume of moisture which is expected to leak into the waste through the cover clay liner. Several models are currently available to simulate the performance of the landfill cover liner. However, these have been shown to underestimate the actual leakage volumes. The existence of macro-pores, especially attributed to desiccation cracking, is hypothesized as the cause of this discrepancy.

Although there have been previous attempts to model flow through soil macro-pores, especially in agricultural engineering, the simplifications employed have rendered the models insufficient for the clay liner problem. Most importantly, previous investigators have bypassed a theoretically robust hydraulics theory encompassing macro-pore flow. Instead, previous models have been phenomenological in nature, and are primarily site-specific.

The conceptual model of flow through cover liners that has been presented acknowledges the existence of macro-pores. Flow through the clay matrix would be modeled using Darcy's Law, while the macro-pore flow component would be modeled using the equations governing analogous processes in open channel and closed conduit hydraulics. These equations are well developed and verified in other applications. Future investigations will provide the complete mathematical model of flow through clay liners incorporating the macro-pore concepts. Collection of field data for verification of the model is underway.

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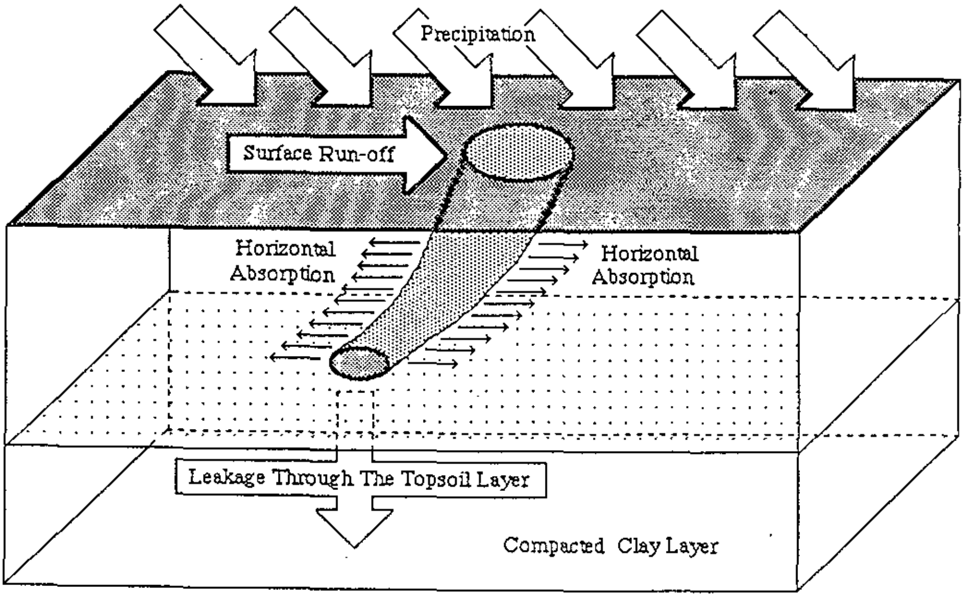

Figure 1: Leakage through the circular conduit type macro-pores in the topsoil layer of a clay cover liner during the rainfall.

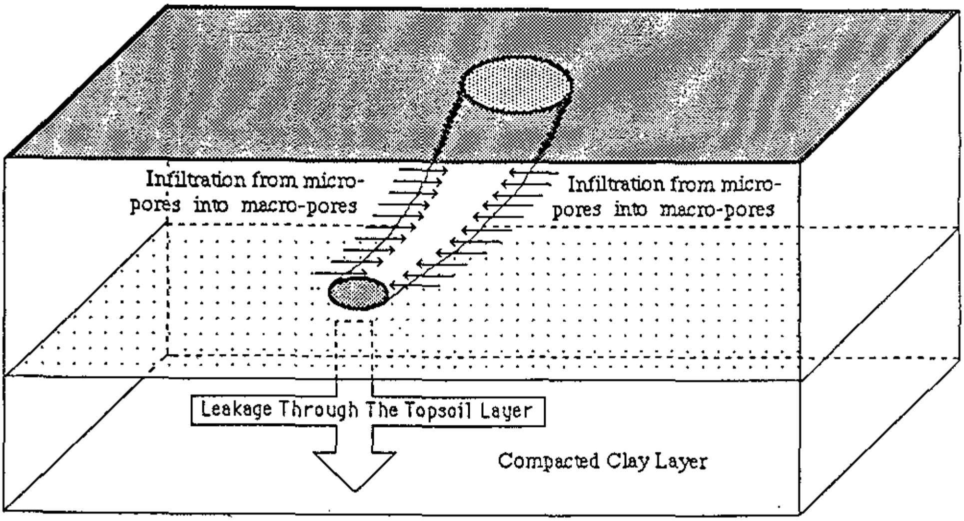

Figure 2 : Postrainfall leakage through the circular conduit type macro-pores in the topsoil layer of a clay cover liner.

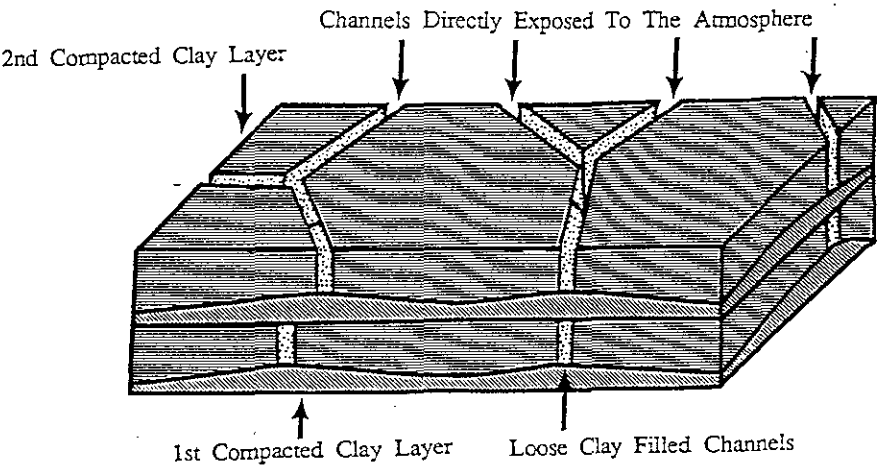

Figure 3: Flow Through Multiple Layers of Cracked Clay Liner Exposed at the Surface

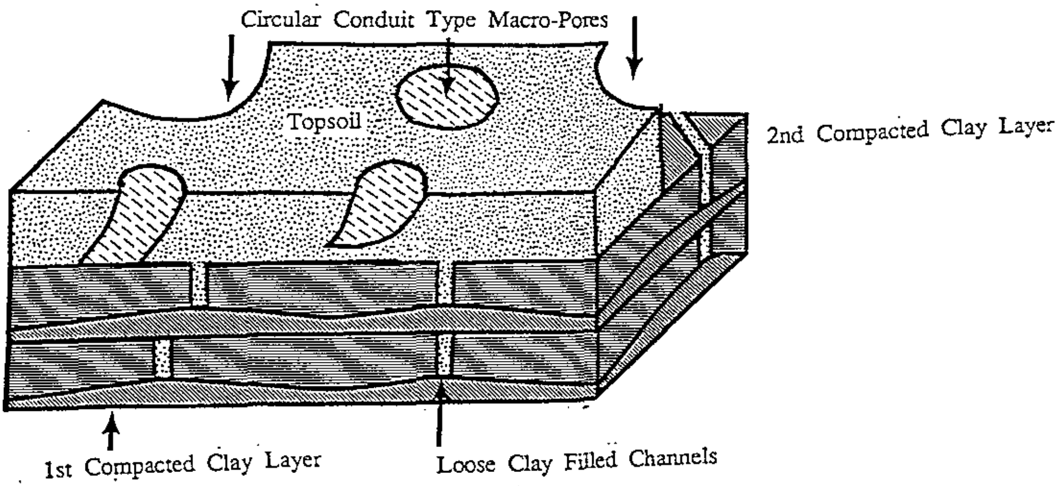

Figure 4: Flow Through a Combination of Macro-Pores in the Topsoil and Clay Cover Liner

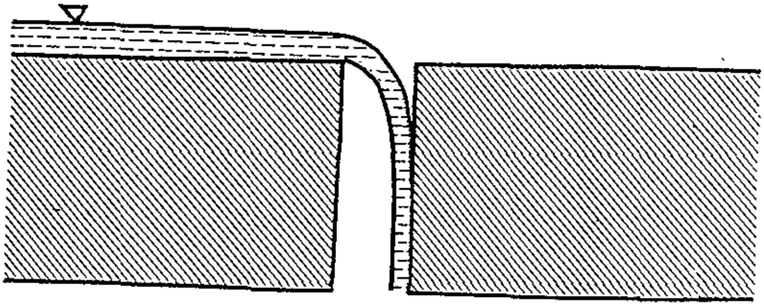

Figure 5: Free-falling weir flow into cracks exposed at the surface

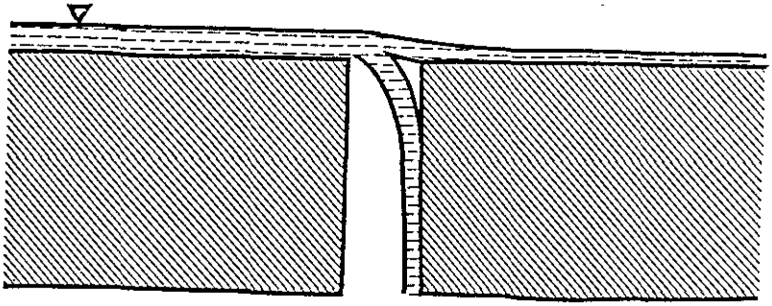

Figure 6: Flow partitioning in cracks exposed at the surface

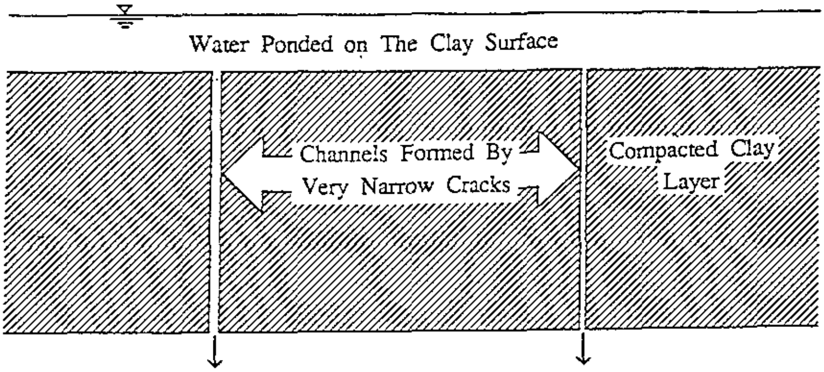

Figure 7: Flow through water-filled cracks of clay liner

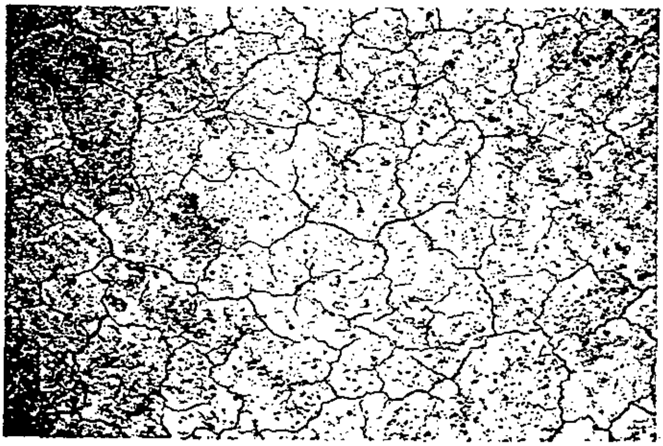

Photo 1: Desiccation cracks distributed throughout clay cover.

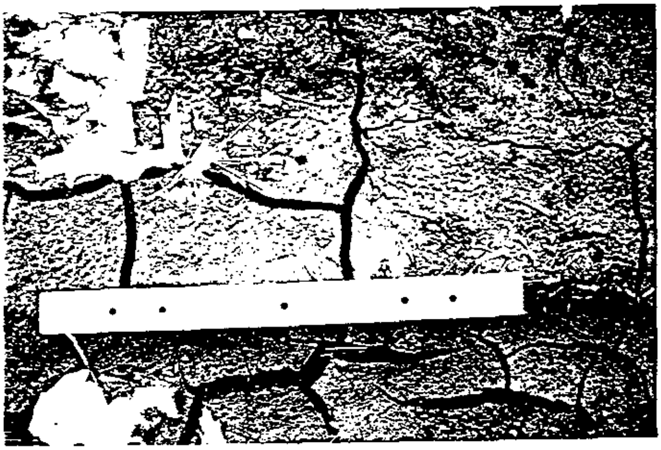

Photo 2: Desiccation cracks on clay cover have been observed at widths exceeding 1 cm. For scale comparison, a 30.5 cm straight-edge appears in the photo.

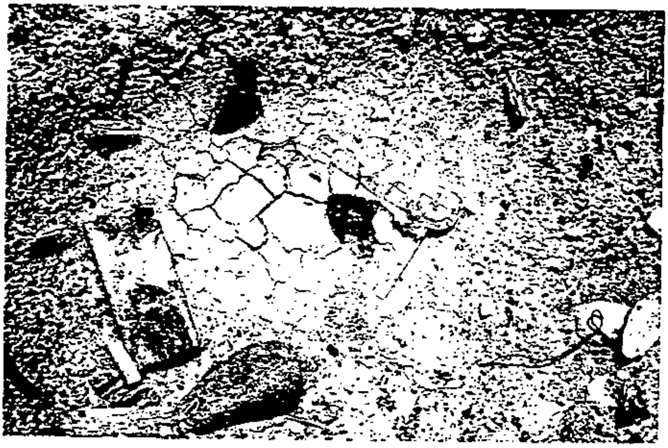

Photo 3: The clay ped removed from the clay layer can be clearly seen.

TWO-YEAR WATER BALANCE MEASUREMENTS OF THE CAPILLARY BARRIER TEST FIELD AT BØTTERUP, DENMARK. PRELIMINARY RESULTS.

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B. Nilsson.
Geological Survey of Denmark.

Summary

The paper briefly describes the capillary barrier test site at Bøtterup, North Zealand, its design, construction and equipment. Data collection for a period from April 1986 to June 1988 are reported. The data comprises precipitation, leakage through the barrier, and barrier-drainage. Water balance calculations are tabulated and given graphically for succeedingly increasing periods. The results are discussed and the paper concludes that the capillary barrier from the two most sloping sections over a length of 40 m were able to drain off 42% of 1510 mm precipitation over a period of 799 days.

INTRODUCTION

Prevention of groundwater pollution may be achieved by decreasing the leachate from waste disposal facilities.

A simple way of doing this is to install a top cover above the waste to decrease the infiltration and recharge of water into the waste and to increase the interflow within the covering layer.

By using the capillary barrier as a top cover the interflow is increased, and infiltrated water is drained outside the barrier-covered area before it reaches the waste.

The capillary barrier concept is mentioned for the first time by Rançon (Rançon 1972), and experiments in laboratory and field on a smaller scale is carried out by Rançon (Rançon 1972 and 1974) and Madsen (Madsen 1983). Full-scale experiments were first performed in Denmark at Bøtterup (Andersen & Madsen 1985), and the preliminary results up to January 1987 from these experiments were reported by Andersen and co-workers in 1985, 1986 and 1988. The present paper gives the results up to July 1988.

Modelling of two-dimensional flow applicable to the capillary barrier is published by Jensen et al. (1986).

Field tests on a larger scale have been started in Germany, (Melchor, 1987) and France, (Barres et al. 1988).

DESIGN AND FUNCTION OF THE CAPILLARY BARRIER

The capillary barrier consists of two layers of permeable and porous sand. A fine-grained one above and a coarse-grained one below. The interface between the layers is sloping. Recharging water will be stored in the fine sand and will move according to the gradient. The coarse layer will remain dry as long as tensional forces are present in the fine layer. In this way the capillary barrier can lead recharging water laterally away and prevent creation of leachate from wastes when placed above the waste.

THE PRINCIPLE OF THE CAPILLARY BARRIER

The principle of the capillary barrier (figure 1) is based on the fact that fine-grained

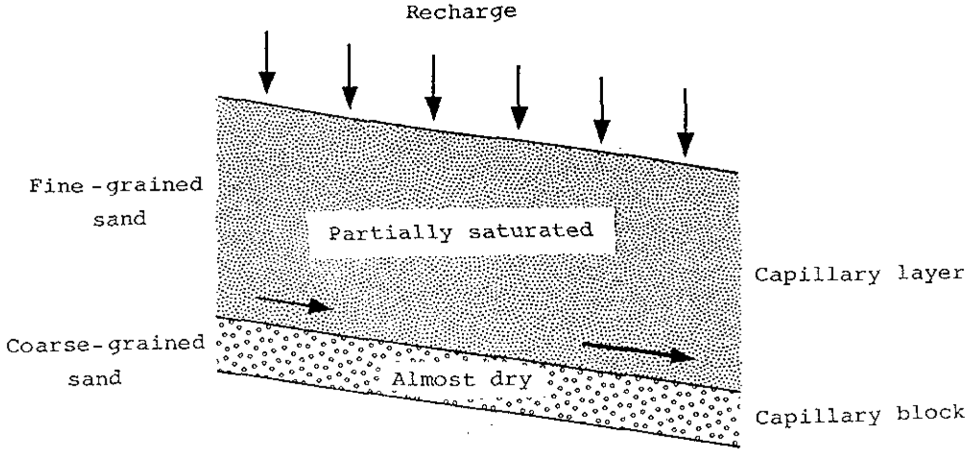


Figure 1.: Sketch of the principle of the "Capillary Barrier". (After Andersen & Madsen 1985).

sand under unsaturated conditions contains more water and has a higher hydraulic conductivity than coarse-grained sand. A coarse-grained sand material superimposed by a fine-sand material will, under recharging conditions, build up a capillary zone in the lower part of the fine sand. With a sloping interface between the two layers the water within the capillary fringe of the fine sand will move in the direction of the slope. The hydraulic conductivity of the capillary zone of the fine sand is great, in relation to that of the coarse sand below the interface. A lateral flow will occur in the capillary zone, and only little water will penetrate the interface and move downwards into the coarse sand.

THE FULL-SCALE TEST FIELD AT BØTTERUP

A full-scale test field was designed and constructed at Bøtterup, North of Copenhagen, Denmark, during 1985, as described by Andersen and Madsen (1985) and Andersen and Christiansen (1986). The test field consists of 5 areal sections, see figures 2 and 3. Each section is 15 by 20 m, placed in series with increasing slopes of 2, 4, 6, 8 and 10 %. The whole test field is surrounded by a drain trench, and each section has a transversal drain trench adjoining the neighbouring section for collection of leakage. The whole test field and the surrounding ditch and transversal trenches are covered with a plastic liner to prevent interference with water in the surroundings. In each section are installed two layers of sand. The upper one, the fine-grained, is 60 cm thick and the lower one 30 cm.

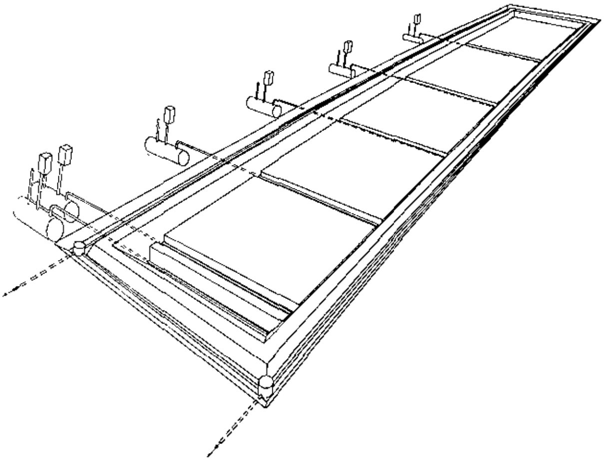


Figure 2.: Bird's eye view of the geometry of the full-scale test site for the capillary barrier experiments at Bøtterup, Denmark.

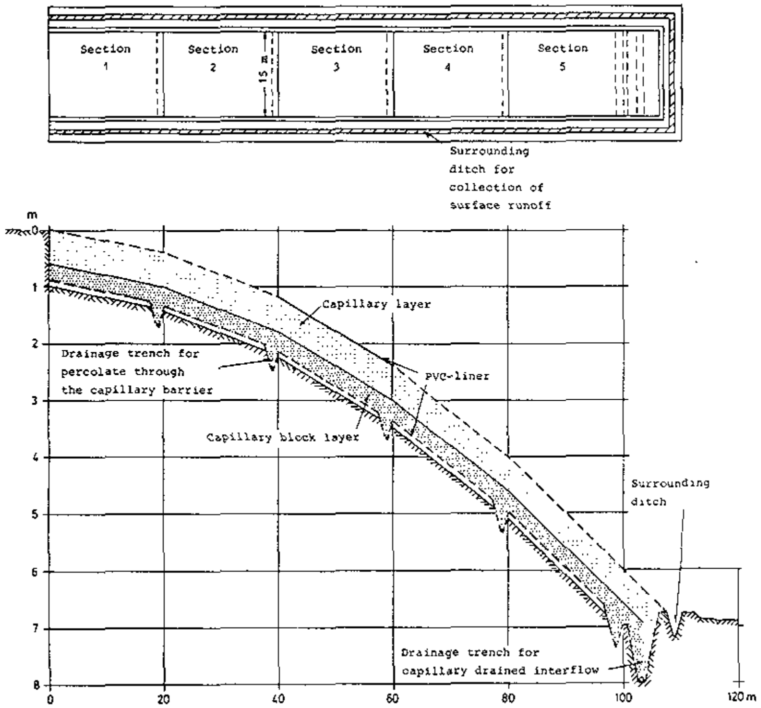
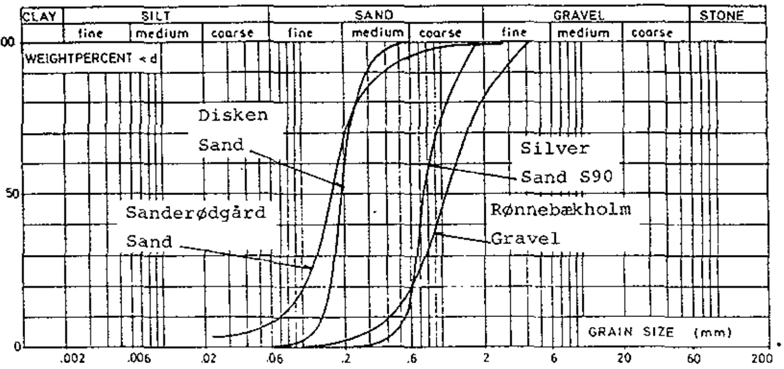


Figure 3.: Plane and vertical view of the capillary barrier facility at Bøtterup, Denmark. (After Andersen & Madsen, 1985).

THE BARRIER MATERIALS

Disken sand and Sanderødgaard sand is used as the capillary upper layer and Silver sand and Rønnebækholm gravel as the coarse bottom layer. The grain-size distribution of the materials and their hydraulic properties appear from figure 4.



Indication	Disken Sand	Sanderødgård Sand	Silver Sand S90	Rønnebækholm Gravel
$U = d_{60}/d_{10}$	1,42	2,30	1,57	2,98
d_{50} (mm)	0,19	0,14	0,77	1,05
h_c (cm)	~45,0	~50,0	~15,0	~25,0
K_{sat} (m/sec)	$1x5x10^{-4}$	$5x10^{-5}$	$1,1x10^{-3}$	$5x10^{-3}$ (est.)

Figure 4.: Grain-size distribution and hydraulic properties of the barrier material at Bøtterup, Denmark. (After Andersen & Madsen, 1985).

OBSERVATION PERIOD

The test fields have been exposed to precipitation from April 24th 1986 till July 1st 1988, i.e. during a period of 799 days. Data from this period are presented and discussed in this paper.

DATA AQUISITION AND DATA COLLECTION

Precipitation, (P) leaked water from each section (QZ1, QZ2, QZ4 and QZ5) and barrier-drained water QO2 and QO5 were measured, the precipitation once a day and the collected water continuously by means of recording equipment in the collecting tanks.

The observed values of the above mentioned parameters are given for each subperiod in table 1 as litres, in table 2 in millimetres, and in table 3 accumulated as millimetres. Finally in table 4 the values are accumulated and given in percentages of precipitation.

RESULTS

The results can be divided into two separate parts, those from sections 1 & 2 and those from sections 4 & 5. Section 3 was covered by a plastic liner, and tank 3 works as collector for barrier-drained water from section 2. There is a vertical separation as a plastic membrane between sections 3 & 4, and therefore there is no flow of water across this boundary.

Sections 1 & 2

In table 3 the accumulated values of leakage and barrier-drained water from sections 1 & 2 are listed for each subperiod.

As for the total period of observation, 799 days, the barrier-drainage was 165 mm, and the leakage 802 mm from a precipitation of 1510 mm. The precipitation figures were corrected for wind effect as indicated in table 1. At the end of the period in percentages of precipitation the barrier-drainage is 11%, the leakage 53%, and the difference between the precipitation and the outflows should be equal to change in stored water and evaporation. As the collection of barrier-drained water passes section 3 before it is collected, the observed values for barrier-drained water may be delayed. This will be discussed below.

Sections 4 & 5

From table 3 appears that the total period of observation was 799 days. Accumulated precipitation corrected for wind effect was 1510 mm. The leakage was 614.6 mm in section 4, and 848.8 mm in section 5, and the barrier has drained 667.4 mm off over a distance in flow direction of 40 m and over a width of 15 m equalizing an area of 600 sqm. The slope of the barrier layer was 8% for section 4, and 10% for section 5.

From figure 5 appears that the barrier-drainage starts within the first month and reaches full capacity, ca. 30 mm per month, within two months.

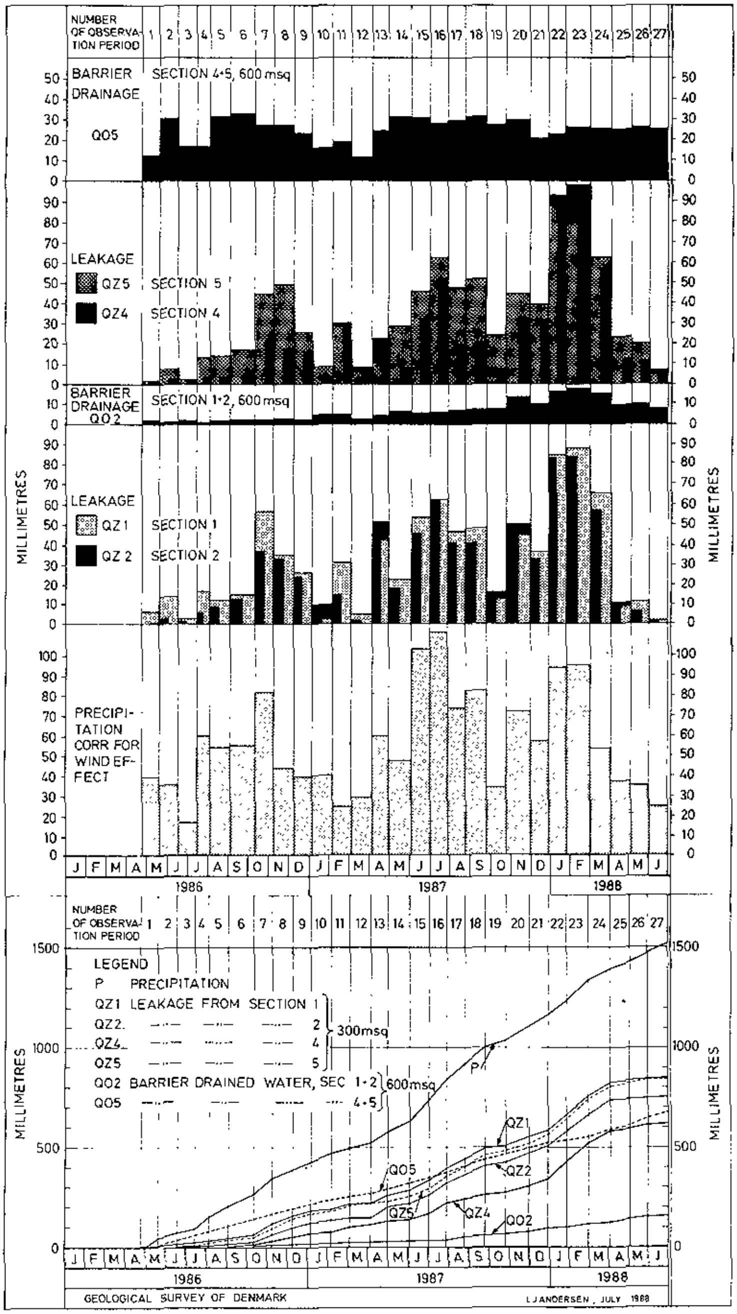


Figure 5.: Periodical and accumulated water balance data for the capillary barrier at Bøtterup April 1986 to June 1988.

The decrease in barrier-drainage as well as leakage from all sections for the period December 1986 to April 1987 is caused by frozen ground with snow accumulation in the test field.

From May 1987 to July 1988 the barrier has drained with nearly full capacity, about 25 mm per month, except for two months with low temperatures, December 1987 and January 1988.

As can be seen from figure 5, the precipitation for 1987 and beginning of 1988 was high. This results in high leakage rates for period no. 22 and 23 nearly equal to the precipitation.

In table 4 the accumulated values are listed in percentages of the precipitation. For the first year more than 50% of the precipitation was drained by the barrier. During the rest of the observation period until June 1988' the barrier-drained water corresponds to 42-48% of precipitation.

In all but two sub-periods the leakage from section 5 was greater than from section 4. The reason for this is that barrier-drained water from section 4 moves into section 5. The transmissivity of the capillary layer in section 5 was not able to transfer both the recharge from section 5 and the barrier-drainage from section 4.

WATER BALANCES

The water balance of the sections 1 & 2 of the barrier can be described by the equation:

$$P = (QZ1+QZ2)/2 - QO2 - EV \pm DS \qquad (1)$$

Where: P : Precipitation
QZ1 : Leakage from sec. 1.
QZ2 : Leakage from sec. 2.
QO2 : Barrier-drainage from sec. 1&2.
EV : Evaporation from sec. 1&2.
DS : Change in stored water within the sections.

As mentioned above the water balance for sections 1 & 2 results in great values of EV and DS. In figure 5 the results are shown graphically as periodical and accumulated values.

Table 5 gives the water balance calculation between the start and the end of each period. The two parameters, EV and DS, evaporation and change in stored water in the capillary barrier, were not measured and therefore the difference between the precipitation, the leakage plus the barrier-drainage, equal to EV + DS, is calculated.

Since the barrier was drained for six months before exposure, the barrier had a low water content at the starting point for the water balance. Therefore the absolute changes under future, natural conditions may be smaller. Since a remarkable part of the changes might be related to change in stored water, DV, the evaporation, EV, from the test field seems to be small.

If a period with the same water content at start and end can be found, the evaporation during such a period could be calculated from the water balance equation for the period, since DS is equal to zero.

It has not been possible to select such periods for sure. However, October 1986 - July 1987 and July 1987 - February 1988 might be times where the barrier has been near maximal storage.

In figure 6 and 7 accumulated values of the various parameters of the water balance is shown graphically, and the lightly hatched area below the graph for the precipitation indicates the size of evaporation and change in storage as a function of time for the whole measuring period.

From the observed values it appears that the sum of change in storage and evaporation is 543.5 mm for sections 1 & 2, and 111 mm for sections 4 & 5. For sections 1 & 2 the values are high and are still increasing, for sections 4 & 5 the values fluctuate seasonally. Therefore the evaporation part of these values might be very small, according to the values from sections 4 & 5, which are better. However, the increasing difference in the values from sections 1 & 2 could indicate a higher evaporation or a leak somewhere. However, there is no reason for a higher evaporation from sections 1 & 2 than from sections 4 & 5, and it has not been possible to discover any leaks by various tests in sections 1 & 2. Therefore the discrepancies between the two observation series might be explained either from a much larger, available storage in section 3, where the barrier-drained water from sections 1 & 2 is collected, or it is caused by damages in the bottom liner under sections 1 & 2. Since the barrier-

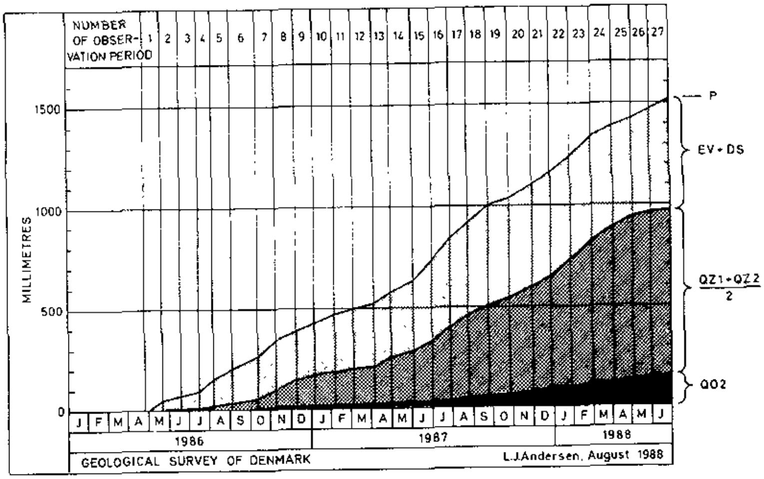


Figure 6.: Accumulated water balance for the capillary barrier sections 1 & 2, Bøtterup, 1986-88.
EV: evaporation, DS: change in storage.
(Other symbols, see figure 5.)

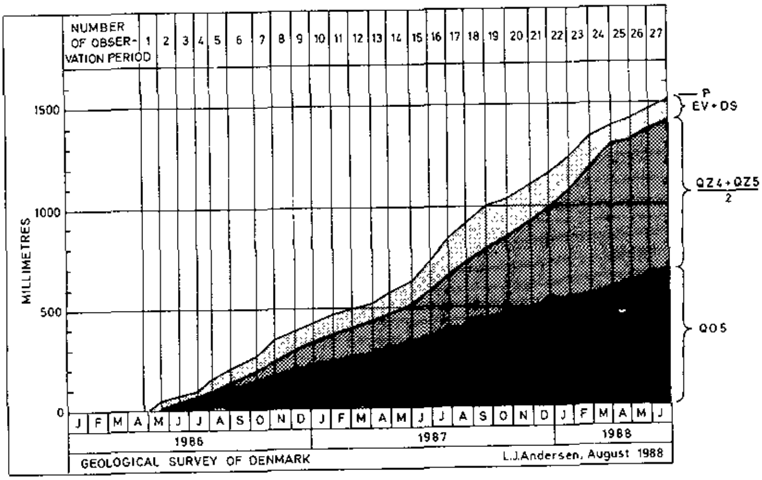


Figure 7.: Accumulated water balance for the capillary barrier sections 4 & 5, Bøtterup, 1986-88.
EV: evaporation, DS: change in storage.
(Other symbols, see figure 5.)

drained water has been increasing during the observation period and since the leakage from sections 1 & 2 and 4 & 5, 801 and 731.7 mm respectively, the low barrier-drainage from sections 1 & 2 seems to be caused partly by a delay of the water in section 3 and partly by a higher evaporation. However, it cannot be excluded that a damage of the bottom liner allows some water to disappear.

Anyhow, the indications for a low evapotranspiration from the barrier seems to be the most reasonable anticipation. From the values from sections 4 & 5, table 5, it seems that the evaporation from these two sections are negligible and not measurable within the measuring accuracy.

FUTURE INVESTIGATIONS

The effect of vegetation and its evapotranspiration are planned to be investigated.

The construction of the barrier will be changed so that it is possible to measure both leakage and barrier-drained water from each section.

CONCLUDING REMARKS

The investigations at the capillary barrier test field at Bøtterup for a period of more than two years have proved that a capillary barrier is able to drain-off remarkable amounts of infiltrated water over reasonable distances.

For sections 4 & 5 44.2% of the precipitation for a period of 799 days were drained off by the barrier. In many periods the barrier-drainage was higher than the recharge.

The investigation periods were rather abnormal, with changes between dry periods and very wet periods, where the monthly precipitation for two succeeding months exceeded 100 m.

During the relative dry period April-June 1988, with a monthly precipitation of about 35 mm the leakage from the barrier is nearly stopped, in June it was as low as one sixth of a mm per day in sections 4 & 5. At the same time the barrier-drainage was about 25 mm per month and still near its maximal capacity. It took about two months to increase the water content in the barrier to a level where maximal drainage capacity was achieved. And only for 3 months, during the winter of 1986-87 with severe frost, the barrier-drainage was lower than 20 mm per month.

There seems to be a rather long response-time regarding the flow rate of the barrier-drained water, and when the storage is filled with water it takes months before the barrier-drainage stops.

Conversely, the leakage rate has a much shorter response-time. After full storage the leakage decreases relatively quickly, when precipitation slows down for longer periods. However, response-time seems to be short during periods of high precipitation and full storage. Another experience is that the decrease in leakage rate appears earlier in a upstream section than in a downstream one.

A uniform recharge of about 200 mm per year should be drainable over flow distances of 50 m and over distances about 25 m double up.

A combination of a low-permeable layer above the barrier to prevent the bigger recharge-rates might result in nearly tight barrier.

Overloading of a downgradient section is often the case.

The evaporation seems to be negligible under the existing conditions with a cover of pebbles.

ACKNOWLEDGEMENT

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Table 1. Values of precipitation, leakage and barrier-drained water at the capillary barrier test site at Bøtterup, Denmark, April 1986 - June 1988.

QZ1,QZ2,QZ4 and QZ5-values refer to leakage from catchment area of one section (300 sqm), while P, Q02 and Q05-values refer to precipitation, barrier-drainage from catchment areas of two sections (600 sqm), respectively.

Observation Period		Number of days	Precipitation	Wind-correction	Precipitation	Leakage		Drainage	Leakage		Drainage
						Sec. 1	Sec. 2	Sec. 1+2	Sec. 4	Sec. 5	Sec. 4+5
			uncorr.		corr.						
NO	DATES	ND	P litres	WC %	P litres	QZ1	QZ2 litres	Q02	QZ4	QZ5 litres	Q05
	1986										
1	24/04-20/05	26	20880	+15	24012	1846	84	739	312	141	7296
2	20/05-19/06	30	18840	+14	21478	3958	628	493	757	2217	18520
3	19/06-15/07	26	9480	+13	10712	592	450	862	348	432	9961
4	15/07-05/08	21	32400	+12	36288	4942	1660	493	700	3831	9821
5	05/08-04/09	30	29400	+11	32634	3593	2483	985	941	3983	19081
6	04/09-11/10	37	29520	+13	33358	4438	3660	1724	1564	4978	19642
7	11/10-09/11	29	42660	+15	49059	16864	11280	1478	6549	13461	16555
8	09/11-10/12	31	22440	+17	26255	10591	10122	1970	5246	14707	16275
9	10/12-06/01	28	19740	+20	23688	7918	6978	1232	4722	7522	14030
	1987										
10	06/01-06/02	31	20040	+21	24248	781	2561	2463	1040	2480	9821
11	06/02-05/03	27	12000	+22	14640	9008	4373	2463	8998	8798	11505
12	05/03-06/04	31	14640	+21	17714	1302	510	1232	2349	1983	6734
13	06/04-01/05	25	30540	+18	36037	12822	15282	2217	6593	3115	14732
14	01/05-03/06	33	24900	+15	28635	6835	5403	3695	2369	8520	18520
15	03/06-03/07	30	54420	+14	62039	16046	13587	3202	9519	13680	18239
16	03/07-29/07	26	59820	+12	66998	18627	18819	3202	15838	18609	16836
17	29/07-26/08	28	39660	+11	44023	14068	12271	3695	5517	14228	17958
18	26/08-25/09	30	44160	+12	49459	14400	12252	3941	6034	15498	18800
19	25/09-26/10	31	17820	+14	20315	3825	4521	4187	2087	7180	16275
20	26/10-01/12	36	37140	+16	43082	13451	14862	7882	10017	13364	17958
21	01/12-29/12	28	29040	+19	34558	10871	9611	5419	9841	11887	12066
	1988										
22	29/12-26/01	28	46560	+21	56338	25922	25142	10098	27933	26263	13749
23	26/01-29/02	34	46860	+22	57169	26543	25203	11084	29661	24680	15714
24	29/02-03/04	34	26760	+21	32380	19563	17296	9113	17087	18848	14872
25	03/04-03/05	30	19560	+18	23081	3137	3335	5419	3400	7007	14311
26	03/05-01/06	29	19260	+15	22149	3138	2048	5665	3138	5744	15994
27	01/06-01/07	30	13740	+14	15664	781	766	3941	1826	1486	15152

Table 2. Precipitation, leakage and barrier-drained water, in millimetres, at the capillary barrier test site at Bøtterup, Denmark, April 1986 - June 1988. in millimetres.

QZ1,QZ2,QZ4 and QZ5-values refer to leakage from a catchment area of *one section (300 sqm), while Q02 and Q05-values* refer to barrier-drained water from a catchment area of two sections (600 sqm), respectively.

Collecting tank no:				1	2	3	4	5	6
Observation Period		Number of days	Precip. 1)	Leakage Sec. 1	Leakage Sec. 2	Drained water Sec. 1+2	Leakage Sec. 4	Leakage Sec. 5	Drained water Sec. 4+5
NO	DATES	ND	P [mm]	QZ1 [mm]	QZ2 [mm]	Q02 [mm]	QZ4 [mm]	QZ5 [mm]	Q05 [mm]
	1986								
1	24/04-20/05	26	40.0	6.15	0.28	1.23	1.04	0.47	12.16
2	20/05-19/06	30	35.8	13.19	2.09	0.82	2.52	7.39	30.87
3	19/06-15/07	26	17.9	1.97	1.50	1.44	1.16	1.44	16.60
4	15/07-05/08	21	60.5	16.47	5.53	0.82	2.33	12.77	16.37
5	05/08-04/09	30	54.4	11.98	8.28	1.64	3.14	13.28	31.80
6	04/09-11/10	37	55.6	14.79	12.20	2.87	5.21	16.59	32.74
7	11/10-09/11	29	81.8	56.21	37.60	2.46	21.83	44.87	27.59
8	09/11-10/12	31	43.8	35.30	33.74	3.28	17.49	49.02	27.12
9	10/12-06/01	28	39.5	26.39	23.26	2.05	15.74	25.07	23.38
	1987								
10	06/01-06/02	31	40.4	2.60	8.54	4.11	3.47	8.27	16.37
11	06/02-05/03	27	24.4	30.03	14.58	4.11	29.99	29.33	19.17
12	05/03-06/04	31	29.5	4.34	1.70	2.05	7.83	6.61	11.22
13	06/04-01/05	25	60.1	42.74	50.94	3.69	21.98	10.38	24.55
14	01/05-03/06	33	47.7	22.78	18.01	6.16	7.90	28.40	30.87
15	03/06-03/07	30	103.4	53.49	45.29	5.34	31.73	45.60	30.40
16	03/07-29/07	26	111.7	62.09	62.73	5.34	52.79	62.03	28.06
17	29/07-26/08	28	73.4	46.89	40.90	6.16	18.39	47.43	29.93
18	26/08-25/09	30	82.4	48.00	40.84	6.57	20.11	51.66	31.33
19	25/09-26/10	31	33.9	12.75	15.07	6.98	6.96	23.93	27.12
20	26/10-01/12	36	71.8	44.84	49.54	13.14	33.39	44.55	29.93
21	01/12-29/12	28	57.6	36.24	32.04	9.03	32.80	39.62	20.11
	1988								
22	29/12-26/01	28	93.9	86.41	83.81	16.83	93.11	87.54	22.92
23	26/01-29/02	34	95.3	88.48	84.01	18.47	98.87	82.27	26.19
24	29/02-03/04	34	54.0	65.21	57.65	15.19	56.96	62.83	24.79
25	03/04-03/05	30	38.5	10.46	11.12	9.03	11.33	23.36	23.85
26	03/05-01/06	29	36.9	10.46	6.83	9.44	10.46	19.15	26.66
27	01/06-01/07	30	26.1	2.60	2.55	6.57	6.09	4.95	25.25

1) corrected for windeffect *(see table 1)*.

Table 3. Accumulated number of days, precipitation, leachate and barrier-drained water in mm, at the capillary barrier test site at Bøtterup, Denmark, April 1986 - June 1988.

QZ1,QZ2,QZ4 and QZ5-values refer to leakage from a catchment area of one section (300 sqm), while Q02 and Q05-values refer to barrier-drainage from a catchment area of two sections (600 sqm), respectively.

Period		Accum. Number of days	Accum. Precipitation	Accum. Leakage		Accum. Drainage	Accum. Leakage		Accum. Drainage
				Sec. 1	Sec. 2	Sec. 1+2	Sec. 4	Sec. 5	Sec. 4+5
NO	DATES	SUM ND	SUM P mm	SUM QZ1 mm	SUM QZ2 mm	SUM Q02 mm	SUM QZ4 mm	SUM QZ5 mm	SUM Q05 mm
	1986-								
1- 1	24/04-20/05	26	40.0	6.2	0.3	1.2	1.0	0.5	12.2
1- 2	24/04-19/06	56	75.8	19.3	2.4	2.1	3.6	7.9	43.0
1- 3	24/04-15/07	82	93.7	21.3	3.9	3.5	4.7	9.3	59.6
1- 4	24/04-05/08	103	154.2	37.8	9.4	4.3	7.1	22.1	76.0
1- 5	24/04-04/09	133	208.5	49.8	17.7	6.0	10.2	35.3	107.8
1- 6	24/04-11/10	170	264.1	64.6	29.9	8.8	15.4	51.9	140.5
1- 7	24/04-09/11	199	345.9	120.8	67.5	11.3	37.2	96.8	168.1
1- 8	24/04-10/12	230	389.7	156.1	101.2	14.6	54.7	145.8	195.3
1- 9	24/04-06/01	258	429.1	182.5	124.5	16.6	70.5	170.9	218.6
	-1987								
1-10	24/04-06/02	289	469.6	185.1	133.0	20.7	73.9	179.2	235.0
1-11	24/04-05/03	316	494.0	215.1	147.6	24.8	103.9	208.5	254.2
1-12	24/04-06/04	347	523.5	219.4	149.3	26.9	111.8	215.1	265.4
1-13	24/04-01/05	372	583.5	262.2	200.2	30.6	133.7	225.5	290.0
1-14	24/04-03/06	405	631.3	285.0	218.3	36.7	141.6	253.9	320.8
1-15	24/04-03/07	435	734.7	338.4	263.5	42.1	173.4	299.5	351.2
1-16	24/04-29/07	461	846.3	400.5	326.3	47.4	226.1	361.5	379.3
1-17	24/04-26/08	489	919.7	447.4	367.2	53.6	244.5	409.0	409.2
1-18	24/04-25/09	519	1002.1	495.4	408.0	60.1	264.7	460.6	440.5
1-19	24/04-26/10	550	1036.0	508.2	423.1	67.1	271.6	484.5	467.7
1-20	24/04-01/12	586	1107.8	553.0	472.6	80.3	305.0	529.1	497.6
1-21	24/04-29/12	614	1165.4	589.3	504.7	89.3	337.8	568.7	517.7
	-1988								
1-22	24/04-26/01	642	1259.3	675.7	588.5	106.1	430.9	656.3	540.6
1-23	24/04-29/02	676	1354.6	764.1	672.5	124.6	529.8	738.5	566.8
1-24	24/04-03/04	710	1408.5	829.3	730.1	139.8	586.7	801.4	591.6
1-25	24/04-03/05	740	1447.0	839.8	741.2	148.8	598.1	824.7	615.4
1-26	24/04-01/06	769	1483.9	850.3	748.1	158.2	608.5	843.9	642.1
1-27	24/04-01/07	799	1510.0	852.9	750.6	164.8	614.6	848.8	667.4

Table 4. Accumulated precipitation, leakage and barrier-drained water in percentages of accumulated precipitation, at the capillary barrier test site at Bøtterup, Denmark, April 1986 - June 1988.

Period	Number of days	Accum. Precipitation	Accum. Leakage		Accum. Drainage	Accum. Leakage		Accum. Drainage
			Sec. 1	Sec. 2	Sec. 1+2	Sec. 4	Sec. 5	Sec. 4+5
		mm	%	%	%	%	%	%
1986-								
1- 1 24/04-20/05	26	40.0	15.4	0.7	3.1	2.6	1.2	30.4
1- 2 24/04-19/06	56	75.8	25.5	3.1	2.7	4.7	10.4	56.7
1- 3 24/04-15/07	82	93.7	22.8	4.1	3.7	5.0	9.9	63.7
1- 4 24/04-05/08	103	154.2	24.5	6.1	2.8	4.6	14.3	49.3
1- 5 24/04-04/09	133	208.5	23.9	8.5	2.9	4.9	17.0	51.7
1- 6 24/04-11/10	170	264.1	24.4	11.3	3.3	5.8	19.7	53.2
1- 7 24/04-09/11	199	345.9	34.9	19.5	3.3	10.8	28.0	48.6
1- 8 24/04-10/12	230	389.7	40.1	26.0	3.7	14.0	37.4	50.1
1- 9 24/04-06/01	258	429.1	42.5	29.0	3.9	16.4	39.8	50.9
-1987								
1-10 24/04-06/02	289	469.6	39.4	28.3	4.4	15.7	38.2	50.0
1-11 24/04-05/03	316	494.0	43.5	29.9	5.0	21.0	42.2	51.5
1-12 24/04-06/04	347	523.5	41.9	28.5	5.1	21.3	41.1	50.7
1-13 24/04-01/05	372	583.5	44.9	34.3	5.2	22.9	38.6	49.7
1-14 24/04-03/06	405	631.3	45.1	34.6	5.8	22.4	40.2	50.8
1-15 24/04-03/07	435	734.7	46.1	35.9	5.7	23.6	40.8	47.8
1-16 24/04-29/07	461	846.3	47.3	38.6	5.6	26.7	42.7	44.8
1-17 24/04-26/08	489	919.7	48.6	39.9	5.8	26.6	44.5	44.5
1-18 24/04-25/09	519	1002.1	49.4	40.7	6.0	26.4	46.0	44.0
1-19 24/04-26/10	550	1036.0	49.1	40.8	6.5	26.2	46.8	45.1
1-20 24/04-01/12	586	1107.8	49.9	42.7	7.2	27.5	47.8	44.9
1-21 24/04-29/12	614	1165.4	50.6	43.3	7.7	29.0	48.8	44.4
-1988								
1-22 24/04-26/01	642	1259.3	53.7	46.7	8.4	34.2	52.1	42.9
1-23 24/04-29/02	676	1354.6	56.4	49.6	9.2	39.1	54.5	41.8
1-24 24/04-03/04	710	1408.5	58.9	51.8	9.9	41.7	56.9	42.0
1-25 24/04-03/05	740	1447.0	58.0	51.2	10.3	41.3	57.0	42.5
1-26 24/04-01/06	769	1483.9	57.3	50.4	10.7	41.0	56.9	43.3
1-27 24/04-01/07	799	1510.0	56.5	49.7	10.9	40.7	56.2	44.2

Table 5. Accumulated waterbalance for sec.1 & 2, at the capillary barrier test site at Bøtterup, Denmark, April 1986 - June 1988

Period		Accum. Number of days	Accum. Precipitation	Accum. Leakage			Accum. Drainage	Accum. Evap. + change in storage
				Sec. 1	Sec. 2	Average Sec. 1+2	Sec. 1+2	Sec. 1+2
NO	DATES	SUM ND	SUM P	SUM QZ1	SUM QZ2	SUM (QZ1+Q22)/2.	SUM Q02	SUM EV+DS
			[mm]	[mm]	[mm]	[mm]	[mm]	[mm]
	1986-							
1- 1	24/04-20/05	26	40.0	6.2	0.3	3.2	1.2	35.6
1- 2	24/04-19/06	56	75.8	19.3	2.4	10.9	2.1	62.9
1- 3	24/04-15/07	82	93.7	21.3	3.9	12.6	3.5	77.6
1- 4	24/04-05/08	103	154.2	37.8	9.4	23.6	4.3	126.2
1- 5	24/04-04/09	133	208.5	49.8	17.7	33.7	6.0	168.9
1- 6	24/04-11/10	170	264.1	64.6	29.9	47.2	8.8	208.1
1- 7	24/04-09/11	199	345.9	120.8	67.5	94.1	11.3	240.5
1- 8	24/04-10/12	230	389.7	156.1	101.2	128.7	14.6	246.4
1- 9	24/04-06/01	258	429.1	182.5	124.5	153.5	16.6	259.0
	-1987							
1-10	24/04-06/02	289	469.6	185.1	133.0	159.0	20.7	289.8
1-11	24/04-05/03	316	494.0	215.1	147.6	181.4	24.8	287.8
1-12	24/04-06/04	347	523.5	219.4	149.3	184.4	26.9	312.2
1-13	24/04-01/05	372	583.5	262.2	200.2	231.2	30.6	321.7
1-14	24/04-03/06	405	631.3	285.0	218.3	251.6	36.7	342.9
1-15	24/04-03/07	435	734.7	338.4	263.5	301.0	42.1	391.6
1-16	24/04-29/07	461	846.3	400.5	326.3	363.4	47.4	435.5
1-17	24/04-26/08	489	919.7	447.4	367.2	407.3	53.6	458.8
1-18	24/04-25/09	519	1002.1	495.4	408.0	451.7	60.1	490.3
1-19	24/04-26/10	550	1036.0	508.2	423.1	465.6	67.1	503.2
1-20	24/04-01/12	586	1107.8	553.0	472.6	512.8	80.3	514.7
1-21	24/04-29/12	614	1165.4	589.3	504.7	547.0	89.3	529.1
	-1988							
1-22	24/04-26/01	642	1259.3	675.7	588.5	632.1	106.1	521.1
1-23	24/04-29/02	676	1354.6	764.1	672.5	718.3	124.6	511.7
1-24	24/04-03/04	710	1408.5	829.3	730.1	779.7	139.8	489.0
1-25	24/04-03/05	740	1447.0	839.8	741.2	790.5	148.8	507.7
1-26	24/04-01/06	769	1483.9	850.3	748.1	799.2	158.2	526.5
1-27	24/04-01/07	799	1510.0	852.9	750.6	801.7	164.8	543.5

Table 6. Accumulated waterbalance for sec.4 & 5, at the capillary barrier test site at Bøtterup, Denmark, April 1986 - June 1988.

Period		Accum. Number of days	Accum. Precipitation	Accum. Leakage			Accum. Drainage	Accum. Evap. + change in storage
				Sec. 4	Sec. 5	Average Sec. 4+5	Sec. 4+5	Sec. 4+5
NO	DATES	SUM ND	SUM P	SUM QZ4	SUM QZ5	SUM (QZ4+QZ5)/2.	SUM Q05	SUM EV+DS
			[mm]	[mm]	[mm]	[mm]	[mm]	[mm]
	1986-							
1- 1	24/04-20/05	26	40.0	1.0	0.5	0.8	12.2	27.1
1- 2	24/04-19/06	56	75.8	3.6	7.9	5.7	43.0	27.1
1- 3	24/04-15/07	82	93.7	4.7	9.3	7.0	59.6	27.0
1- 4	24/04-05/08	103	154.2	7.1	22.1	14.6	76.0	63.6
1- 5	24/04-04/09	133	208.5	10.2	35.3	22.8	107.8	78.0
1- 6	24/04-11/10	170	264.1	15.4	51.9	33.7	140.5	89.9
1- 7	24/04-09/11	199	345.9	37.2	96.8	67.0	168.1	110.7
1- 8	24/04-10/12	230	389.7	54.7	145.8	100.3	195.3	94.1
1- 9	24/04-06/01	258	429.1	70.5	170.9	120.7	218.6	89.8
	-1987							
1-10	24/04-06/02	289	469.6	73.9	179.2	126.6	235.0	108.0
1-11	24/04-05/03	316	494.0	103.9	208.5	156.2	254.2	83.6
1-12	24/04-06/04	347	523.5	111.8	215.1	163.4	265.4	94.6
1-13	24/04-01/05	372	583.5	133.7	225.5	179.6	290.0	114.0
1-14	24/04-03/06	405	631.3	141.6	253.9	197.8	320.8	112.7
1-15	24/04-03/07	435	734.7	173.4	299.5	236.4	351.2	147.0
1-16	24/04-29/07	461	846.3	226.1	361.5	293.8	379.3	173.2
1-17	24/04-26/08	489	919.7	244.5	409.0	326.7	409.2	183.7
1-18	24/04-25/09	519	1002.1	264.7	460.6	362.6	440.5	199.0
1-19	24/04-26/10	550	1036.0	271.6	484.5	378.1	467.7	190.2
1-20	24/04-01/12	586	1107.8	305.0	529.1	417.1	497.6	193.1
1-21	24/04-29/12	614	1165.4	337.8	568.7	453.3	517.7	194.4
	-1988							
1-22	24/04-26/01	642	1259.3	430.9	656.3	543.6	540.6	175.1
1-23	24/04-29/02	676	1354.6	529.8	738.5	634.2	566.8	153.6
1-24	24/04-03/04	710	1408.5	586.7	801.4	694.0	591.6	122.9
1-25	24/04-03/05	740	1447.0	598.1	824.7	711.4	615.4	120.2
1-26	24/04-01/06	769	1483.9	608.5	843.9	726.2	642.1	115.6
1-27	24/04-01/07	799	1510.0	614.6	848.8	731.7	667.4	110.9

LARGE-SCALE LYSIMETER STUDIES ON THE WATER BALANCE OF MULTILAYERED COVERING SYSTEMS ON THE WASTE DISPOSAL SITE GEORGSWERDER (HAMBURG, FRG)

- MEASURING CONCEPT AND FIRST RESULTS -

Stefan Melchior and Günter Miehlich

Summary

Six large-scale test fields (50 m * 10 m) have been built on the waste disposal site Georgswerder in Hamburg (FRG) to determine the water balance of several multilayered landfill caps with different barrier systems under varying slope conditions. The design of the test fields, the measuring concept and first results showing the amounts of water collected below the different barrier systems are presented. The data indicate a significant pore water discharge out of the compacted clay layers due to consolidation. Up to now all systems, though not yet under steady-state conditions, perform better than required by regulations. The best results are achieved by an extended capillary barrier system.

INTRODUCTION

Along with the remedial action on the waste disposal site Georgswerder in Hamburg (FRG) a research and development program with regard to the water balance of various covering systems is being carried out. For this purpose six large-scale test fields with a size of 500 m^2 each have been built (Fig. 1). The tests are supported by the Federal Ministry of Research and Technology (BMFT) and the City of Hamburg and focus on the comparative evaluation of the long-term effectiveness of the different covering systems.

DESIGN OF THE TEST FIELDS AND MEASURING CONCEPT

Each test field is 10 m wide and 50 m long in slope direction (Fig. 1). The fields are integrated in the landfill cover. Flow from the high slope and lateral flow into the test fields are prevented by flexible membrane liners (HDPE) and clay seals. By the help of this design, which was developed in cooperation

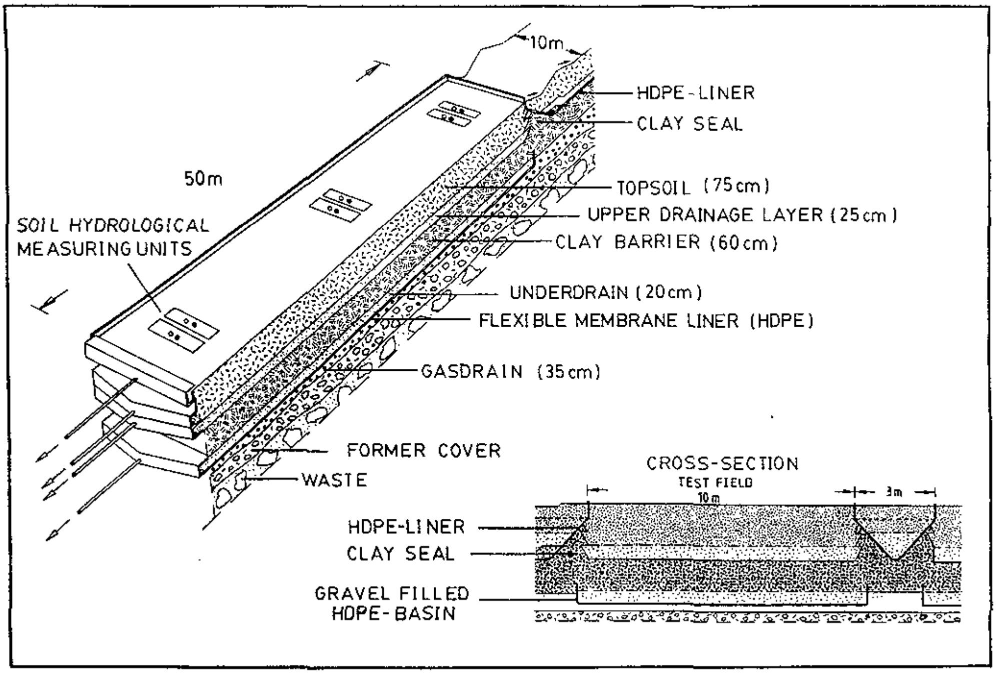

Figure 1: Schematic view on a test field

with the engineering office IGB (Hamburg, FRG) typical lysimeter boundary effects, as described by HÖTZL and WOHNLICH (1987) are avoided and it was possible to construct the clay barrier layers over the whole area under typical construction conditions. Gravel filled underdrains consisting of welded HDPE-liners are installed below the barrier systems. They enable the collection and direct measurement of the barrier system leakage.

The test fields are located in two areas with different slope (F-fields with 4 %, S-fields with 20 % slope). The factors slope and design of the barrier system are varied in such a way that the influence of these factors on the water balance can be determined separately (Fig. 2). All fields are designed as multi-layer-systems with a combination of topsoil, drainage and barrier layers. Due to the comparative testing concept the topsoil and the drainage layer of all fields are designed alike (75 cm of sandy loam respectively 25 cm of fine gravel, separated by a geofabric to protect the drainage layer). The barrier systems vary as follows (Fig. 2):

- A compacted clay liner on F1 and S1 (three lifts of glacial till, 60 cm thick) as often built on disposal sites with relatively low toxic potential.

- A combined barrier system composed of a welded flexible membrane liner (HDPE) on top of a compacted clay layer (F2 and S2) as it

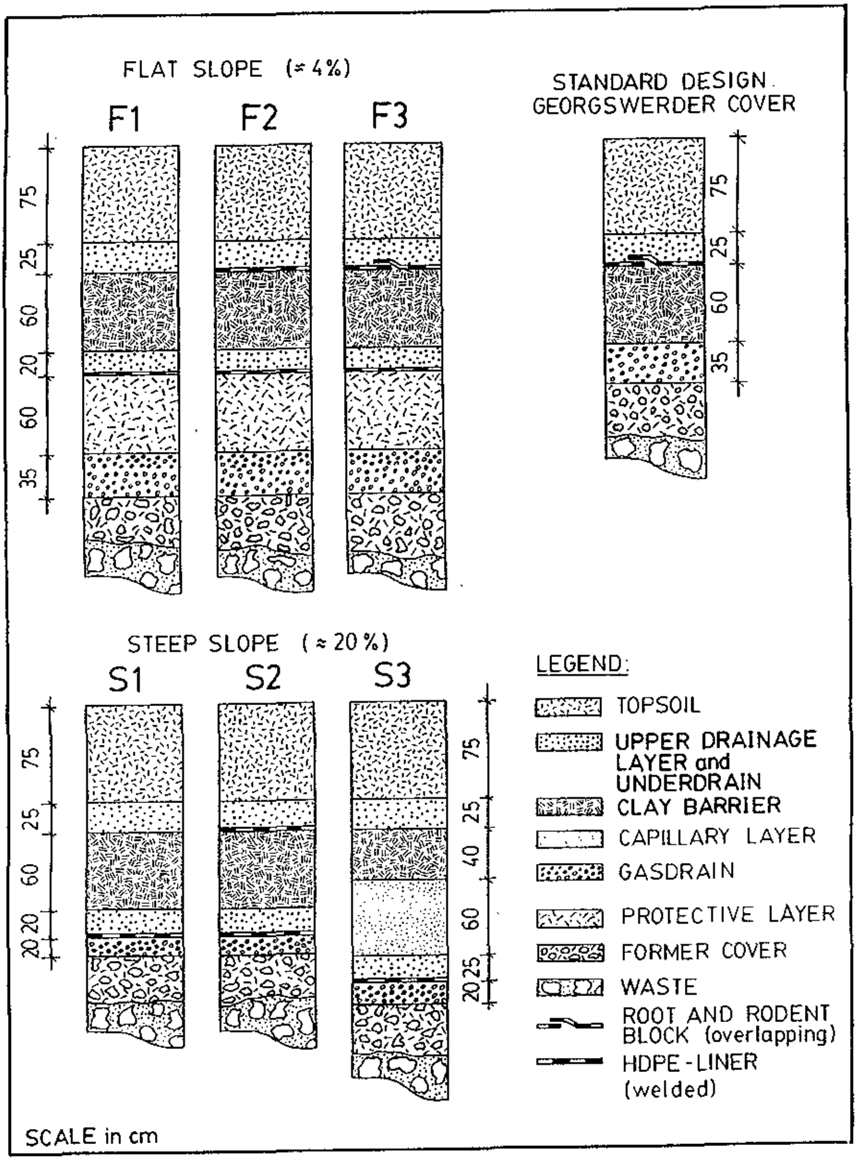

Figure 2: Layer design of the test fields

is commonly used on hazardous landfills. During the test these fields serve as zero plots concerning the percolation of the barrier system, so that the barrier system leakage in the other fields cannot only be evaluated by measuring the discharge in the respective underdrain directly but also independently by the calculus of difference between the upper drainage layer discharges in F2 and S2 and the upper drainage layer discharges in the other variants with respective slope. In addition these fields allow the precise detection of pore water discharge due to consolidation of the clay liners after the installation of upper drainage layer and topsoil.

- Standard design of the Georgswerder cover (F3). Like F2 and S2, HDPE-liners, however, not welded but installed overlapping in slope direction, on top of a compacted clay liner.

- Combined barrier system consisting of a compacted clay layer (40 cm), lying above a capillary barrier consisting of 60 cm capillary layer (fine-grained sand) on top of 25 cm capillary block layer (coarse sand/fine-grained gravel, S3). The upper clay layer provides a low intensive infiltration into the capillary layer even after heavy rainfalls. Therefore the capillary barrier can operate under unsaturated conditions using the wick effect (see ANDERSEN and MADSEN (1985) and CARTWRIGHT et al. (1987)). The aim of this design is to avoid infiltration into the landfill without the use of plastic liners with a yet unknown long-term stability.

The measuring program covers the following range:

- The discharges (surface run-off, interflow in topsoil and drainage layer as well as leakage through the barrier system) are collected and measured individually and with high temporal resolution.

- In some fields various soil hydrological measuring units with a total of 24 neutron probe tubes and 531 automatically recording tensiometers are installed (e.g. in Fig. 1) to define moisture content and matric potential in high temporal and spatial resolution.

- A weather station serves to measure precipitation, air temperature, humidity, wind velocity, radiation balance and soil temperature to determine the water input by precipitation and all parameters necessary to compute evapotranspiration using the Penman/Monteith equation.

In addition to this several soil physical, chemical and mechanical parameters are analyzed in the laboratory or determined by additional experiments in the field: (e.g. grain-size and pore-size distribution, saturated and unsaturated hydraulic conductivity, bulk density, proctor density, plasticity, shear strength, clay mineralogical composition, inorganic constituents of the discharges, tracer and sealed double-ring infiltrometer experiments to determine preferential flowpaths and transit times).

OBJECTIVES AND FIRST RESULTS

The research and development program has started in 1986. After one year of planning the test fields were built in 1987. Two years of intensive measurements followed by a long-term monitoring are planned. The research focusses on different aspects with regard to methodology and objectives:

- Formation of annual water balances for the comparative evaluation of the effectiveness of the test field variants (minimization of infiltration into the waste, suitability of the cover to maintain vegetation, long-term resistance of topsoil, drainage and barrier system towards loads and stresses due to desiccation, erosion, root and rodent penetration, clogging and subsidence).
- Studies regarding the physics of water movement in compacted clay layers with and without covering geomembranes and across layer boundaries with different inclination (validity of DARCY-equation, effectiveness of hydraulic gradients, influence of slope).
- Application of numeric simulation models for reasons of variation and generalized illustration of the influence of different factors (slope, layer design and thickness, soil physical properties, vegetation cover and development, meteorological parameters) as well as for the development of uncomplicated estimating procedures and models for the water balance of covering systems.

The measurements to determine the water balances of the fields (precipitation, discharges and soil water contents) started in the end of last year. Therefore only a very preliminary description of the systems can be given. They show the following phenomena:

- There is surface run-off under both slope conditions, though frequency and intensity is higher on the S-fields. Only very little erosion occurred during winter even though there was almost no vegetation on the fields.
- Lateral flow within the topsoil only occurs on the S-fields. It contributes only a few percent to the total balance.
- The widest range of flow intensities with a strong correlation to rainfall events can be observed in the upper drain layers. There seems to be only little effect of slope concerning the total volume of flow but the flow intensity can be more than two times higher on the steep slope.
- Fig. 3 shows the amounts of water that have been collected underneath the barrier systems in liters per week. The data show a significant pore water discharge out of the compacted clay layers due to consolidation during the construction of upper drain layer and topsoil. Some weeks after the end of construction the discharges decrease. Since there is still pore water discharge (as can be seen on F2 and S2 which have impervious flexible membrane liners on top of the clay barrier) and therefore no steady state conditions have been established yet, no values for the hydraulic conductivity of the clay barriers on F1 and S1 can be calculated out of these data. At least it can be said that there is percolation through the clay barriers of these fields in addition to the pore water discharge because the water volume collected on F1 and S1 is significantly higher than on the respective fields F2 and S2. The higher flow rates on the F-fields, compared with the S-fields are not likely an effect of slope but are caused by the higher initial moisture contents of the clay layers of these fields because weather conditions during construction were wetter. Nevertheless on all fields the total water volumes collected (pore water discharge included) are much less than the amounts of water one would expect calculating the leakage through clay barriers assuming a hydraulic conductivity of 1*10-9 m/s which had been required by regulations. Even better results are achieved with the

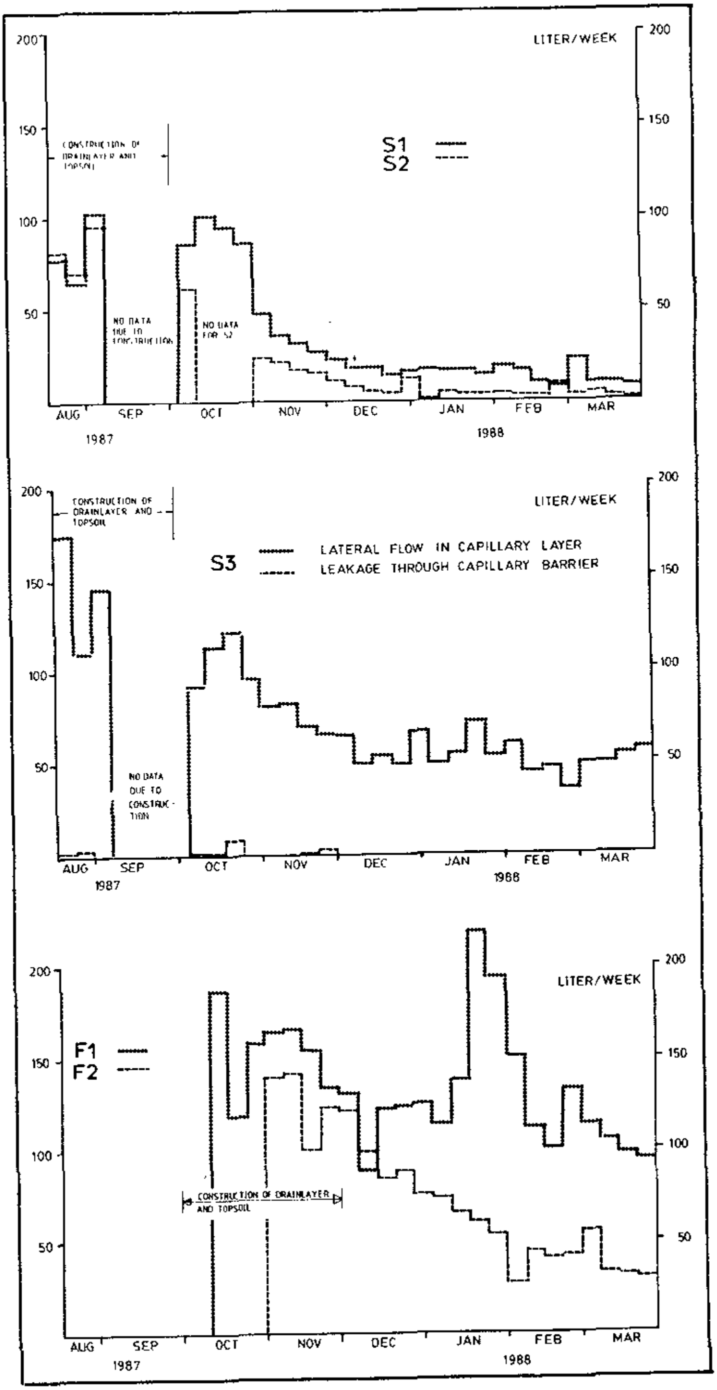

Figure 3: Water collected underneath the different barrier systems (see Fig. 2) in liters per week

extended capillary barrier system on S3. There is more water infiltrating from the clay barrier into the capillary layer as is collected in the underdrains of S1 and S2 (only 40 cm of clay, higher initial moisture content), but nearly all of this water moves laterally within the fine sand and does not pass the capillary barrier.

Further measurements will show how the systems perform on the long run. A more intensive interpretation of the data showing the water balances, response times of the different discharges to rainfall events and of the spatial variability of the soil properties measured during construction is in progress.

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THE MANAGEMENT OF THE HYDROLOGIC SYSTEM OF THE LANDFILL HAMBURG-GEORGSWERDER

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Summary

The main goal of the rehabilitation of the abandoned landfill Hamburg-Georgswerder is to protect the surface waters and the upper, quaternary aquifer against hazardous leachates. The clean up strategy aims to prevent the input of harmful fluids into the ground-water by means of remedial actions, which will result in a change of the hydrologic system of the landfill. A multi-layer cover system and the extraction of the perched water-oil mixture shall dry up the landfill and stop hazardous leachates seeping out of the body of perched fluids through the underlaying marsh sediments into the ground-water.

This paper presents the results of a tentative water balance computation referring to the status of rehabilitation: before, during and after the remedial actions (phase I,II,III). The estimation is based upon precipitation and runoff measurements as well as evaporation calculations. The seepage out of the landfill into the ground-water is roughly calculated by means of the water balance equation. When the rehabilitation will be finished, the remaining seepage into the groundwater will be less than 10 mm per year.

1. INTRODUCTION

The abandoned landfill Hamburg-Georgswerder is situated in the Elbe valley on marsh sediments in the south of Hamburg. From 1948 to 1979, about 14 mio.m³ waste material (containing about 200.000 m³ industrial waste from petrochemical and pesticide industry) has been deposited at this location. In 1983 the highly toxic 2,3,7,8-TCDD (Seveso-dioxine) was detected in the oily leachate.

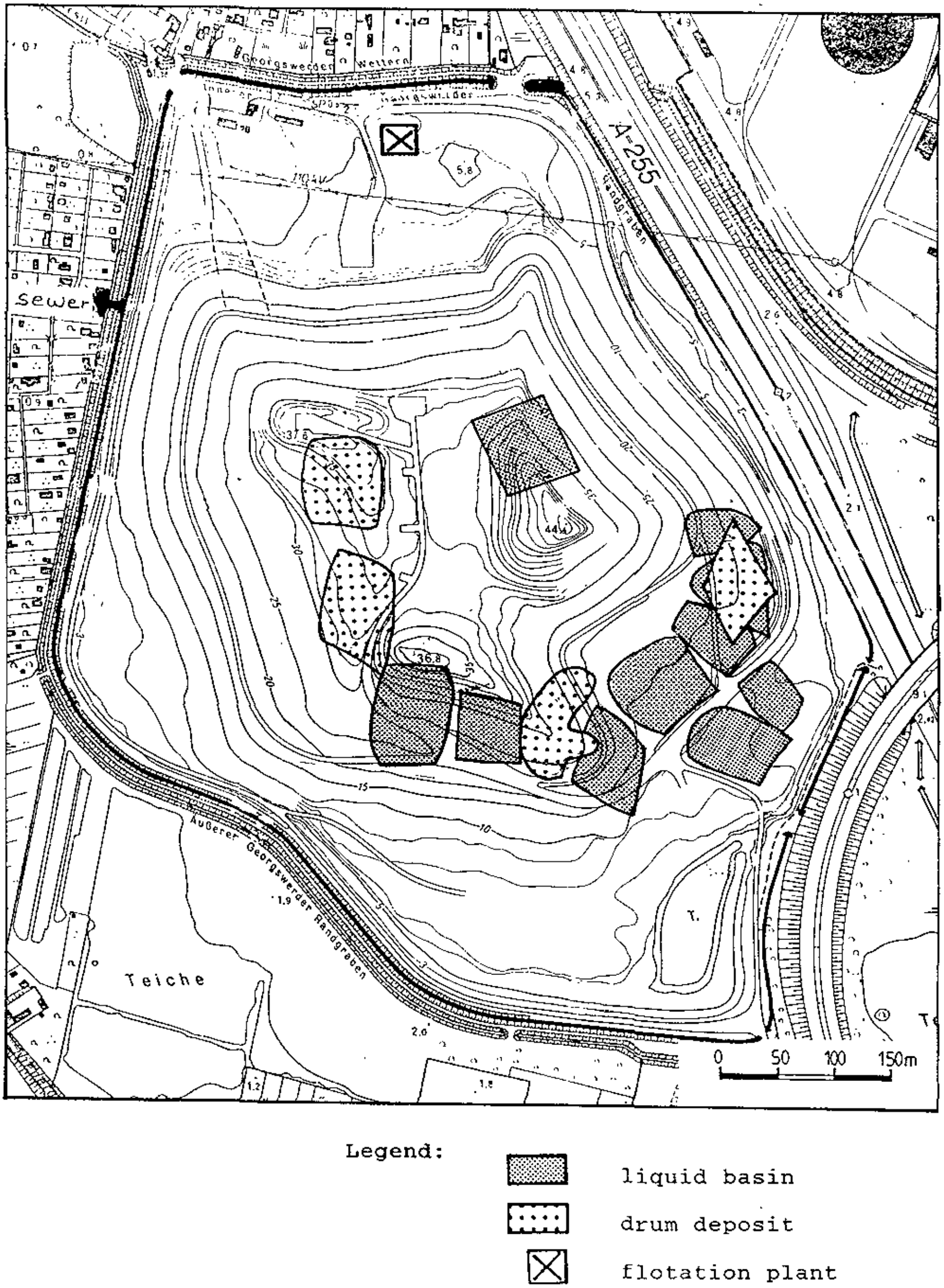

FIGURE 1: Waste disposal site Hamburg-Georgswerder with liquid basins and drum deposits, both containing industrial wastes.

Georgswerder is one of the largest abandoned waste disposal site of Europe, covering an area of 44 ha (figure 1). The rehabilitation program started in 1984. The government has provided 146 mio DM for the rehabilitation (UMWELTBEHÖRDE HAMBURG, 1988). Up to now, among other remedial actions, one third of the landfill has been covered by a multi-layer cover system.

The main goal of all remedial actions is to protect the surface and ground-water against hazardous leachates. So far, the contamination of the near-surface quaternary aquifer is rather low. The limits for drinking water are exceeded slightly only at a few points, where e.g. aromatic solvents, volatile chlororganic solvents and chlorphenoles are found in the ground-water.

The main purpose of the rehabilitation is to prevent the output of harmful fluids into the environment (WOLF, 1988). Remedial techniques are necessary to manage the hydrologic system of the landfill and in particular to manage the large volume of free movable liquids (water-oil-mixture) inside the waste dump.

2. THE HYDROLOGIC SYSTEM OF THE LANDFILL

Figure 2 shows the essential hydrological flow components of the hill-shaped landfill (height: 44 m) at the beginning of the rehabilitation: The precipitation (N) splits into overland flow (A_O), evaporation from vegetation and soil (ET), subsurface flow within the multilayer cover (A_D) and seepage (Z_1) into the unsaturated zone and the body of perched water inside the landfill. The following equation can be set up:

$$Z_1 = N - A_O - ET - A_D \qquad (1)$$

(In all equations changes in reservoir contents and settlement effects are neglected.)

The water-layer inside the landfill forms a coherent perched water table up to 14 m high above the low permeable (natural) bottom layer of the waste dump. In the hillfoot area are numerous outlets of fluids (A_S) from the saturated zone. All lateral flows are collected in a ditch around the landfill. Runoff from surrounding areas (A_X) is part of the total runoff of the ditch (A). Thus the water balance calculation becomes more difficult.

$$A - A_X = (A_O + A_D + A_S) \qquad (2)$$

The *component* Z_2 is most essential for the potential ground-water contamination. Z_2 is the seepage out of the body of perched water through the bottom layer consisting of holocene marsh sediments (clay, peat and mud) into the first aquifer.

$$Z_2 = Z_1 - A_S \qquad (3)$$

The natural sealing system at the base of the waste dump is weakened by former clay extraction and bomb holes.

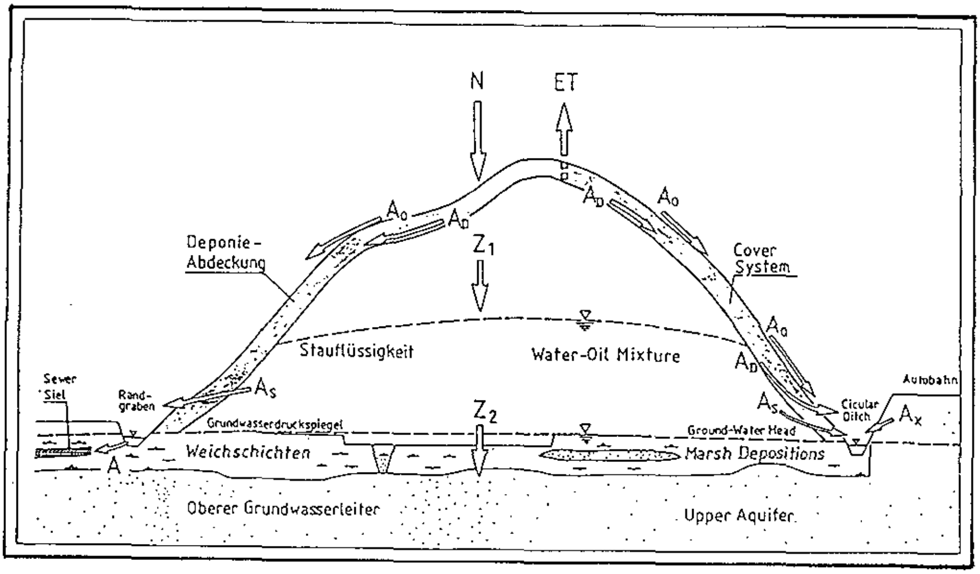

FIGURE 2: The hydrological system of the landfill Hamburg-Georgswerder (schematic cross-section)

3. METHODS TO DETERMINE THE HYDROLOGIC COMPONENTS

Precipitation (N) is measured at 5 sites. Evapotranspiration (ET) is calculated by different methods, e.g.the "HAUDE/soil-water balance" method (SOKOLLEK,1983 ; ERNSTBERGER, 1987) which takes into account vegetation and soil parameters.

During the built of the cover the lateral flow components A_O and A_D are estimated considering the construction phases of the cover system. When it is finished, the total A_O and A_D of the cover system will be measured continuously at 6 points. Thus, equation (1) can be used for a calculation of Z_1.

Six large lysimeters, each 50*10 m, have been built in the cover to investigate the water balance of different types of covers at different inclinations. Measuring started at the end of 1987 (cf. MELCHIOR/MIEHLICH, 1988).

The body of perched water inside the landfill is observed at about 25 points. Some of the monitoring wells form multi-level gauges. At present time, measurements of A_S are carried out at the eastern leachate collecting system. An inductive flux-meter monitores the total runoff of the ditch (A).

The component Z_2, which is the seepage into the ground-water, is estimated by use of different methods:

- Application of the water balance equation:

  $$Z_2 = N - ET - (A_O + A_D + A_S) \qquad (4)$$

- Using a flow model, i.e. application of Darcy's approach, including spatially distributed estimates of the marsh sediment permeability and level differences between perched water table and ground -water table.

- Moreover a more sophisticated hydraulic and transport model will be developed.

4. HYDROLOGIC EFFECTS OF REMEDIAL ACTIONS

In order to reduce the volume (about 1 mio.m^3) of free movable perched water/oil inside the landfill as soon as possible, the following remedial actions are carried out or planned:

- Building a new surface covering, which consists of a grass cover, top substrate (75 cm), drainage layer (25 cm), sealing system (PE-HD-foil above 60 cm compacted marl; $k_f < 10^{-9}$ m/s) and gas drainage (GÜNTHER, 1988).

- Improved collection of the leachates at the hillfoot of the landfill, which are lead to a specific purification plant (flotation plant).

- Forced extraction of waste liquids, first of all from the liquid basins of the landfill (GÜNTHER/MESCHEDE, 1988).

Table 1 shows, how the most important water balance components will change by rehabilitation. The preliminary values are derived from first measurements, expert reports and literature. The calculation takes into account the state of rehabilitation (phase I-III) and the respective pattern of different surface types, as grassland and trees/bushes before rehabilitation, new cover system (in built or complete) and concrete areas.

Evaporation **(ET)** will decrease markedly during the build up of the cover (phase II), owing to a stepwise removal of vegetation. At the final state (phase III) gras/shrub vegetation as well as concrete areas will build the surface. Then ET will be lower than in phase I.

The component A_0 may reach high values temporarily in phase II, e.g. when the low permeable layer (marl) is built on large areas. The subsurface flow A_D becomes the greatest flow component besides ET, when the new cover is finished. On the other hand the seepage into the landfill Z_I will decrease from ca. 190 mm/a in phase I to a very small amount.

In the beginning of phase II, the liquid removal (A_S) from the waste dump by improved collection and forced extraction will reach a higher value than in phase I, limited by the purification plant capacity. Later on, when the perched water table gets lower, A_S will decrease exponentially.

So far, the quantification of the seepage into the groundwater Z_2 is most uncertain. The presented estimation shows, that the remedial actions will reduce the leachate seeping into the aquifer to a very small amount.

PHASE		ET	A_O	A_D	Z_1	A_S	Z_2
I Before rehabilitation:							
permeable cover, various vegetation types	mm:	500	10	0	190	140	50
	1000 m^3:	220	4	0	84	62	22
- circa 1982 to 1986 -							
II During remedial actions (Szenarium B):							
One third of the landfill covered, one third cover in built;collection and extraction of leachates	mm:	350	210	100	40	200	30
	1000 m^3:	155	90	45	20	90	13
- circa 1990 -							
III After rehabilitation:							
new cover finished, most of mobil perched water/oil removed	mm:	430	90	160	20	10	10
	1000 m^3:	190	40	70	9	5	4
- after circa 2000 -							

TABLE 1: Preliminary values of the main water balance components of the landfill Hamburg-Georgswerder at different phases, given in mm/a or 1000 m^3/a during "normal" years with N = 700 mm/a or 308.000 m^3/a.

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ENVIRONMENTAL CONTROL AT DISPOSAL SITES BY REDUCING THE LEAKAGE RATE

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SUMMARY

Reduction of the leakage rate from disposal sites is a convenient method for environmental control at disposal sites when it comes to waste from which contaminants are released over a long period of time. The leakage rate can be reduced by appropriate measures when the deposit is constructed, and no maintenance or further control should be needed in the future.

To achieve the required reduction of the leakage rate the deposit should preferably be sited in an inflow area for groundwater thus preventing penetration of groundwater through the deposit. The deposit should also be covered with a sealing layer, if necessary a drainage layer above, and a final cover of soil to protect the underlaying layers from frost and erosion.

If groundwater inflow is prevented then the leakage rate equals the infiltration of precipitation through the sealing layer. Calculations based on long-term climatic observations and laboratory studies indicates that the leakage rate can be estimated as the product of the saturated hydraulic conductivity of the sealing layer and the total time during an average year when groundwater formation takes place. In addition, reduction of the leakage rate caused by a drainage layer can be estimated to 75% of the calculated leakage rate.

This method of estimating leakage rate has not yet been verified in field studies. However, an evaluation of the disposal technique and of the dimensioning calculations will be made in connection to the covering of two deposits with mine dumps in southern Sweden. The effects of the covering on the water balance of the disposal area are included in the extensive environmental monitoring programme. The first results from the monitoring should be available in the autumn of 1989.

INTRODUCTION

The technique used in most disposal sites (expecially local authority refuse tips) in Sweden today is based on collection and treatment of the leachate. After treatment, the leachate is released into a recipient. Normally, the leachate is sent to a sewage treatment plant, where it is processed together with other municipal sewage. This technique was developed when the dominant type of refuse was biologically degradable household refuse and the leachate problem focussed on BOD and to some extent also nutrient salts. As the disposal sites become full, they are covered over with ordinary soil which allows plants to become established. Previously, the leachate treatment was often terminated when the site was no longer used or some time after.

In recent years, however, it has been observed that this disposal technique is not especially suitable for much of the refuse now produced. This applies, for example, to combustion wastes, which produce leachates contaminated with non-degradable substances, principally salts and trace elements. Such leachates are directly unsuitable for conventional sewage treatment plants. In addition, large quantities of wastes are produced by Swedish industries extracting copper from sulphidic ores. Serious environmental problems have appeared in this connection owing to weathering (oxidation of the sulphides) followed by the formation of sulphuric acid and leaching of heavy metals. Also much of the refuse desposited at municipal sites today gives rise to leachates containing non-degradable or very slowly degradable substances, both organic and inorganic. Another important characteristic of this type of refuse is that leaching takes place over a very long period of time (thousands of years).

The occurrence of wastes which are new for Swedish conditions, together with the publicity given to the special environmental problems of mining wastes, for example, has led to the development of a "new" strategy for disposal of such wastes. Work on this model has mainly been carried on in governmentfunded studies of the environmental consequences of combustion of coal, peat and household refuse for energy production, and in the project "Waste deposits in the mining industry", funded by the National Swedish Board of Environment Protection, see for example (1), (2), (3), and (4).

DISPOSAL STRATEGY

The proposed disposal strategy involves a "controlled leaching" of the contaminants in wastes, i.e. removing contamination at such a slow rate that the surrounding environment is unaffected and so that no water sources are polluted. This means that the increase in the background contents of various substances in surface and ground water must not become excessive. The controlled leaching should be achieved by action at the disposal stage and when ending disposal. No further maintenance should be required. In view of the long time during which leaching normally takes place, it is unrealistic to consider action that would involve regular attention and maintenance in the future.

In most cases, action to reduce the contents of contaminants in the leachate produced are both diffucult and expensive to implement. The measures normally considered are thus a restriction of the leachate flow from the deposit. This restriction can be achieved by reducing the inflow of water to the deposit in two ways:

- a suitable localization and design of the deposit, which prevents the entry of ground water and surface water.
- a suitable covering of the deposit, which limits infiltration by precipitation.

Localization is of great importance. If the deposit is localized to an inflow area for ground water, this effectively eliminates the possibilities of ground water penetrating the deposit, see Fig. 1. An effective localization and design of the deposit with regard to the surrounding topography also prevents the penetration of run-off water. Localization to an inflow area also causes the leachate to leave the deposit as an addition to the ground water, provided that leachate formation does not exceed the infiltration capacity of the ground. Often this is an advantage, since part of the contaminants are fixed in the ground beneath the deposit and contents in the leachate are thereby reduced.

For this reason, the deposit should be localized high up in the run-off area, close to the ground watershed. The contaminants then travel long distances in soil and rock providing good possibilities for fixing and dilution before they reach any ground water or surface water aquifers of importance. Alternatively, the deposit can be positioned close to a suitable recipient with a large water turnover where good dilution is immediately obtained.

Figure 1 also shows the general effects of different measures that can be considered for reducing leachate formation. It can be seen that the best effect is obtained if use is made of surface barriers combined with localization to inflow areas. Localization to outflow areas may sometimes be preferable for other reasons (reclamation of land, "non-hazardous" waste, suitable recipients). Where large quantities of penetrating ground water occur, bottom sealing may also be considered. When sited to inflow areas, bottom sealing of the deposit does not affect the amount of leachate produced, but can instead affect whether the leachate is released to surface water or to ground water, as can be seen from figure 1.

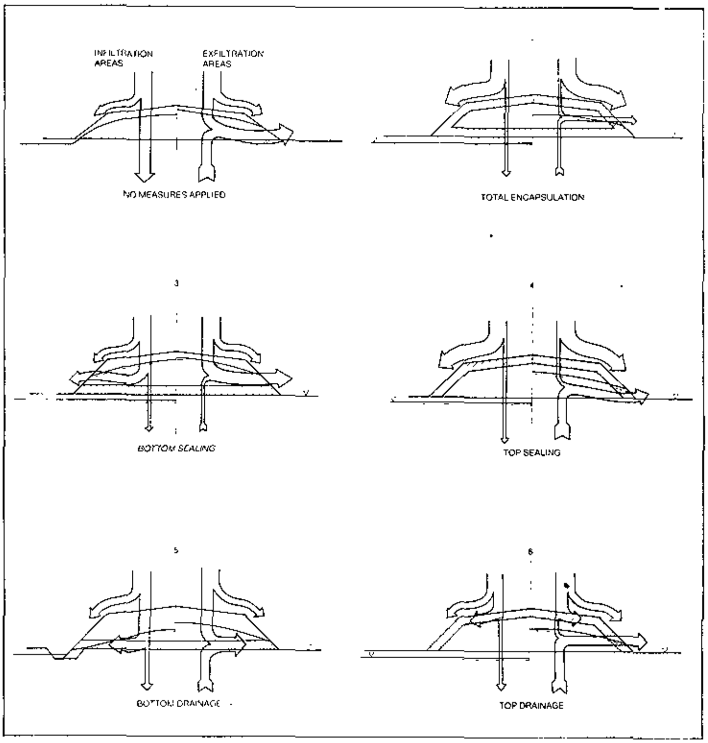

Figure 1. General influence on the water balance in a deposit according to the type of site and measures (1).

DIMENSIONING THE COVERING LAYER

The cover on the deposit, which is designed to reduce infiltration by precipitation, can be designed in different ways. The general model proposed in Sweden, e.g. for combustion residues and mining wastes from sulphidic ores, is shown in figure 2.

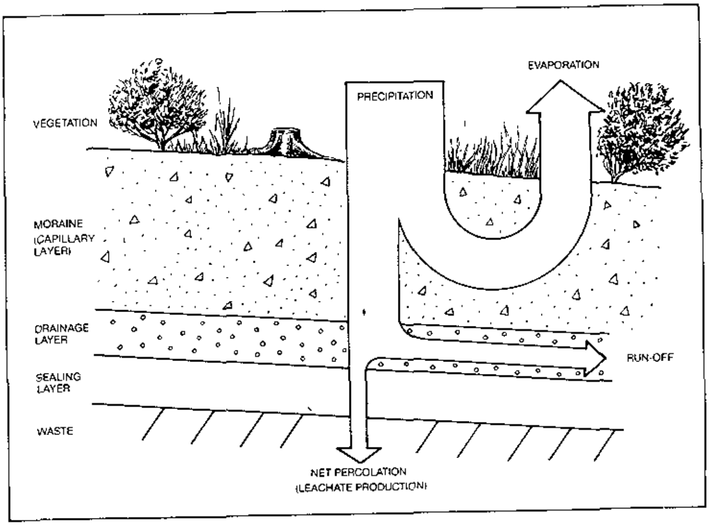

Figure 2. General function of the suggested cover (3).

The sealing layer must have sufficiently low permeability for the desired reduction in infiltration by precipitation to be obtained. A covering capillary layer provides a good depth of soil for plant growth, at the same time as evaporation is encouraged. The intervening drainage layer is favourable with regard to the capillary barrier effect. This means that infiltrating water is retained by the capillary process in the cover available for evaporation until the field capacity of the material is exceeded and water can percolate through the drainage layer to the sealing layer. The drainage layer also offers good run-off possibilities for percolating precipitation. The length of time that water is available for percolation through the sealing layer can thus be reduced considerably.

A simple method for estimating percolation by precipitation through a covering layer as described above, i.e. leachate formation, has been developed. This relates to long-term conditions when equilibrium has been established and is based on three assumptions, i.e.-

- percolation (net) through the sealing layer occurs only during those parts of the year when ground water formation (net) occurs in natural ground.

- the hydraulic conductivity of the sealing layer during these periods can be set equal to the saturated hydraulic conductivity (conservative assumption),
- the gradient driving percolation through the sealing layer can be set equal to unity.

The last two assumptions involve in principle an assumption that the sealing layer is water-saturated during the percolation periods. This appears reasonable in view of the fact that the sealing layer consists of fine material with good water retention properties and in addition is protected against drying-out by the overlying cover.

On the above assumptions, the leachate formation during one year can be calculated as:

$$L = K \cdot f_p \cdot t \cdot 10^3 \text{ mm/year}$$

where K = saturated hydraulic conductivity (m/s)

L = leachate flow (mm/year)

f_p = the percolation time factor

t = $3.154 \cdot 10^7$ s (i.e. one year)

A thorough inspection of the fluctuations in the ground water level based on long-term observations at a large number of places in Sweden has been made by Nordberg (5). He found that on the basis of these fluctuations, Sweden can be divided into four regions with respect to periods of rising ground water level (net inflow) and falling ground water level (net outflow) as in figure 3. Using these general fluctuations, the different regions can be assigned percolation factors as in figure 3, varying between 0.4 and 0.7. These factors represent the proportion of a year during which leachate formation, i.e. rising ground water level, occurs in the respective region. The assumption that leachate formation only occurs during periods when the inflow to natural ground water aquifers exceeds the outflow is naturally not fully correct. However, corresponding results can also be obtained by processing long-term observations of climatic factors (precipitation, evaporation, run-off, snow accumulation and snow melting). After processing, such data show good agreement with the percolation time factors according to figure 3. Calculations made by the Swedish Meterological and Hydrological Institutet (SMHI) with the HBV model for 3 run-off areas in different parts of Sweden also show good agreement with results obtained from the equation above.

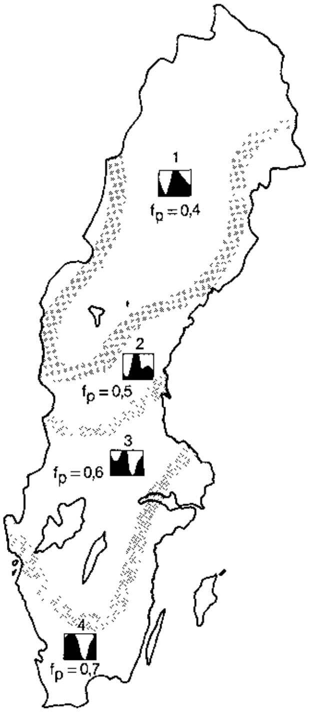

Figure 3. Generalized groundwater fluctuations in different regions of Sweden and assigned percolation time factors (3).

The percolation time factors given in figure 3 apply for a cover consisting of an overlying cover layer (protective layer) of moraine, for example, with relatively good water retention. If the path for capillary transport from protective cover to sealing layer is broken by a drainage layer, the percolation time factors must be reduced. A laboratory study (6) has shown that this reduction can be calculated at about 75%. These experiments were carried out with the aid of an infiltration box in which different covering layers were built up. Standardized rain with varying intensity and duration was then applied to the surfaces with a rain simulator. Typical precipitation periods with high ground water formation were also simulated, using a program developed on a stochastic basis.

A summary of the results in a dimensioning diagram can be made as in figure 4. This diagram is intended to be used together with figure 3, from which the percolation time factor without a drainage layer is determined. If a drainage layer is used, this percolation time factor is *multiplied by 0.25.* Using this dimensioning factor, it is possible to set demands on the cover with regard to maximum permissible hydraulic conductivity, based on the size of the leachate release that can be accepted.

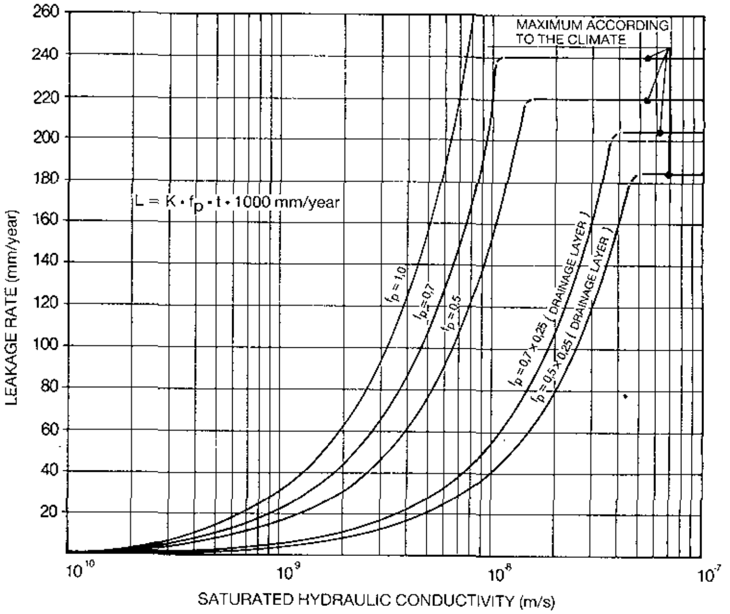

Figure 4. Leakage rate as a function of saturated hydraulic conductivity and percolation time factor (3).

CASE STUDIES

The model described have generally been developed on the drawing board and to some extent through laboratory studies and have not yet been tested in the field. However, covering of two deposits with sulphide mine tailings is now in progress at Bersbo near Åtvidaberg in southern Sweden. In addition to reducing the quantity of leachate, the purpose of this covering is to drastically limit penetration by oxygen to the tailings so as to stop the weathering processes which are accompanied by the release of heavy metals. One of the deposits is being provided with a 0.5 m covering layer of clay compacted in three layers. The other is being given a 0.25 m sealing layer of macadam in a matrix of Cefill. Cefill is the trade name of pozzolanic fly ash, activated with cement and mixed with water and certain other additives into a liquid slurry. In Bersbo this slurry is pumped into a layer of macadam so that it fills the pores. After hardening, the construction forms a concrete-like shell. By varying the mixing procedure Cefill with different, low hydraulic conductivity can be obtained.

The demand on the sealing layers is that the saturated hydraulic conductivity must not exceed $k = 10^{-9}$ m/s. Both the sealing layers will be covered with 2 m moraine. No drainage layer will be used. The thickness of the moraine is motivated by the need to protect the sealing layer against both frost heave and penetration by roots in the future.

The effects of these coverings will be followed up over a long period of time through different types of measurements. These will include both direct tests of the sealing layer's permeability to oxygen and to water and indirect measurements to determine a water balance for the deposits and changes in transport of heavy metals from the deposits. The first results that can provide a guide as to the efficiency of the sealing are expected to be obtained during autumn 1989.

The installations required for the follow up are made during the building up of the coverings. Measurements of the water balance are planned to be carried through with the aid of lysimeters, filter tips for permeability measurements and neutron scattering for measuring water content. The lysimeters, five on each of the deposits, shall be placed directly below the sealing layers in order to collect percolating water from an area of 1 m^2 each. The amount of collected water can then be related to measured permeabilities in the sealing layers. Permeability measurements shall be carried through with the aid of filter tips placed in the sealing layers. Through these filter tips a small amount of water can be forced into the sealing layers by means of a pressure, and the permeability (after saturation) can be calculated. The relationship between the amount of collected water in the lysimeters and the measured permeability can then be used to evaluate the described method of estimating leakage rate. As the water content through the covering will be measured by means of neutron scattering it will also be possible to relate the penetration of water through the sealing layer to the water content in the covering.

In addition, also the precipitation and evaporation at the deposit areas will be measured, as well as the discharge from the deposit area.

During the projecting of the reclamation in Bersbo field tests were performed with the construction of three sealing layers with covering, about 10 x 20 m^2 each. The tested sealants were layers consisting of Cefill and clay as described above and a layer of 0.3 m bentonit mixed in sand (8-11% bentonit). Permeability tests on the sealing layers showed that all the layers fulfilled the demands on low permeability ($\leq 10^{-9}$ m/s). For economical and practical reasons sealing layers of Cefill and clay were chosen for the reclamation.

The permeability tests that have been performed are of a relatively small scale, as ring infiltrometer tests (ø 100-300 mm) and laboratory tests on cores (ø 100 mm) have been used. From other studies it has been shown that permeabilities measured in a small scale especially in laboratory tests sometimes can differ a lot from the over-all permeability of a sealing layer (7). However, in the same studie it was found that ring infiltrometer tests compares favourably to the overall field-measured K. The coverage of the deposits in Bersbo and the extensive follow-up measures will thus be the full-scale test on the practical construction and the function of sealing layers as well as on the dimensioning.

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ON SITE TREATMENT OF SOIL AND GROUNDWATER POLLUTED WITH CHLORINATED ORGANIC COMPOUNDS - A CASE FROM SKRYDSTRUP

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Summary

The present paper gives results from the treatment of soil and groundwater polluted with chlorinated solvents.

In soil the degradation of trichloroethylene has been shown to take place under anaerobic conditions, while degradation under aerobic conditions proceeds very slowly. For 1,1,1-trichloroethane degradation can not be shown conclusively over the time period of the experiments (0 - 15°C, 18 months).

For groundwater clean-up the pumping yield has been optimized by application of the USGS-MOC model. It has been shown that pumping from partially penetrated wells in the central part of the plume is most efficient, and that the configuration of the pumps is most important during the first few years of clean-up. The clean-up is estimated to last for 10 - 20 years.

For the water treatment aeration, probably by two or three aerators placed in series, is a reasonably efficient and very simple process. An activated carbon adsorption can where necessary be applied to remove the final traces of chlorinated solvents.

INTRODUCTION

The present paper deals with a chemical waste disposal site at Skrydstrup in Southern Jutland, Denmark. The site is located at a glaciofluvial plain about 1 km west of the Main Stationary line from the Weichsel glaciation as shown in figure 1.

The site was used for disposal of hazardous waste from a refrigerator factory from 1963 to 1974, the waste consisting of a sludge from degreasing baths, paint sludge, plastic, insulation foam etc.

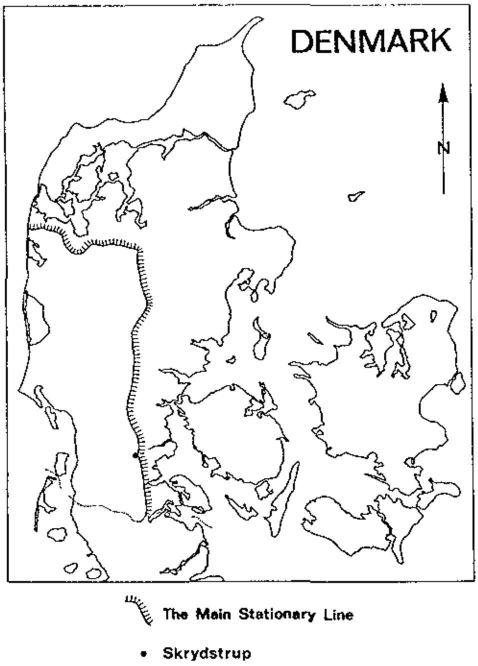

Figure 1 Location of the Skrydstrup site

In 1984 investigations of pollution from the site were initiated. The investigations showed a heavy load of chlorinated solvents and organophosphorous flame retardants in the waste site, as well as a severe pollution of the groundwater below and downstream the site by the same compound types.

In 1986 the site itself was cleared, the chemical wastes removed and incinerated at an intermunicipal chemical waste treatment plant. The contaminated soil was placed in a temporary collection site equipped with heavy polyethylene liners and an efficient leachate collection system. The fate of the chlorinated solvents and the organophosphorous compounds in the site has been followed, and the results are reported here.

A full scale clean-up of the groundwater pollution is planned and will be initiated by the end of 1988.

The dispersion of the contaminated plume has been simulated applying a numerical flow and transport model to the location.

The clean-up is planned on the basis of these simultations and results of a pilot plant study of water treatment processes. The results of these studies are also reported in this paper.

In connection with the full scale clean-up of groundwater 3 research projects have been initiated:

- aerobic biodegradation of chlorinated solvents by the addition of natural gas in columns with activated carbon,
- aerobic biodegration in the unsaturated zone by co-metabolism with methane or propane gas, and
- anaerobic biodegration of chlorinated solvents in contaminated groundwater by the addition of a carbon source.

These projects are now in progress, but it is still too early to give results.

BACKGROUND

Temporary collection site

The temporary collection site is constructed with two separate parts, one having aerobic and the other anaerobic conditions. A cross section of the anaerobic part of the site is shown in figure 2.

The aerobic part of the site has a similar construction without the top liner.

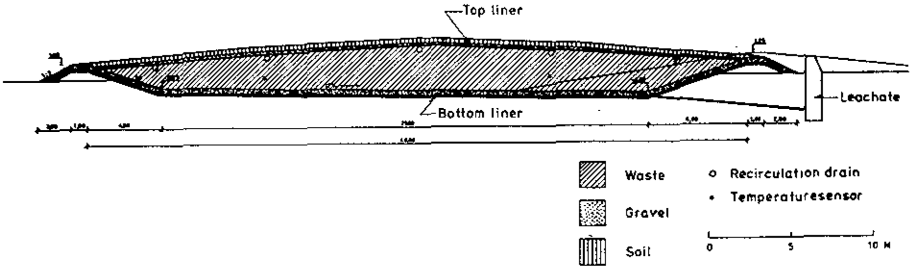

Figure 2 Cross section of the anaerobic part of the temporary collection site

An estimate of the amount of chemicals in the two parts of the site is shown in table 1.

	Aerobic part	Anaerobic part
1,1,1-Trichloroethane	1-2 kg	100 kg
Trichlorethylene	0,1 kg	30 kg
Tetrachloroethylene	<0,1 kg	1 kg
Tris(2-chloroethyl)phosphate	1 kg	4 kg
Tris(3-chloropropyl)phosphate	2 kg	15 kg
Soil	500 m^3	3000 m^3

Table 1 Estimated amount of contaminants in the temporary collection site

The organophosphorous components have shown no degradation in the collection site and thus the following results will only deal with the chlorinated solvents.

Groundwater clean-up

The horizontal extent of the contaminant plume has been determined from measurements of the concentration of the chlorinated organic compounds in bore holes. The results are shown for trichloroethylene and 1,1,1-trichloroethane in figure 3 where also concentration contours from the numerical simulations are shown. The vertical extent of the contaminant plume is determined on the basis of analyses of water samples collected while drilling.

Figure 4 shows the vertical concentration distribution along the plume axis.

To simultate the transient dispersion of solutes from the opening of the waste deposit in 1963 till the present the USGS-MOC model, a well documented and commonly used numerical model, is applied. Input parameters are determined partly from hydrogeological investigations and calibrations partly from the literature what concerns the retardation and degradation of the compounds.

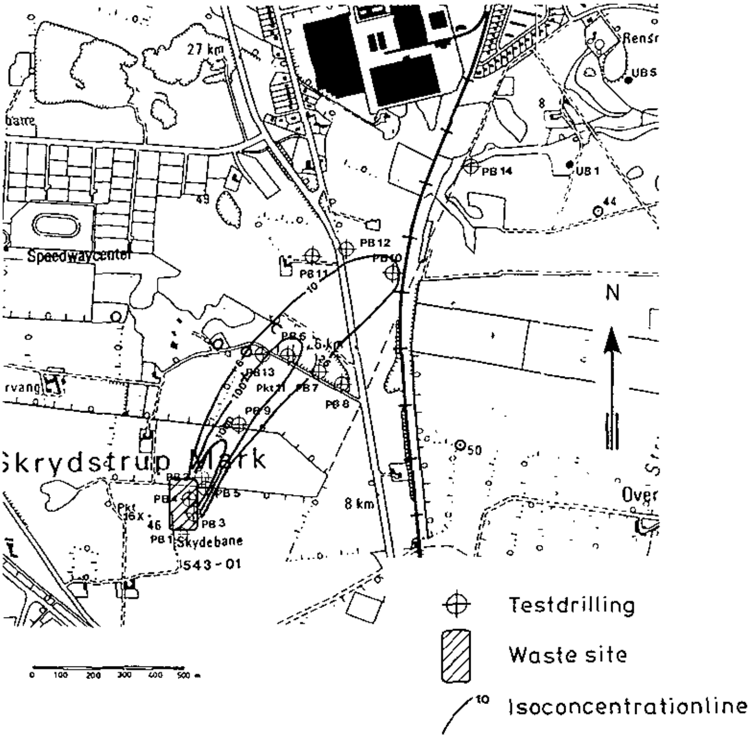

Figure 3 Horizontal distribution of trichloroethylene + 1,1,1-trichloroethane in the groundwater reservoir

The aquifer is a homogeneous, unconfined sand aquifer and its hydrogeological characteristics are shown in table 2.

infiltration	400 mm/year
transmissivity	30×10^{-3} m^2/s
porosity	0,30
longitudinal dispersivity	5,0 m
transversal dispersivity	0,25 m
aquifer thickness	30 m

Table 2 Hydrological characteristics of the aquifer

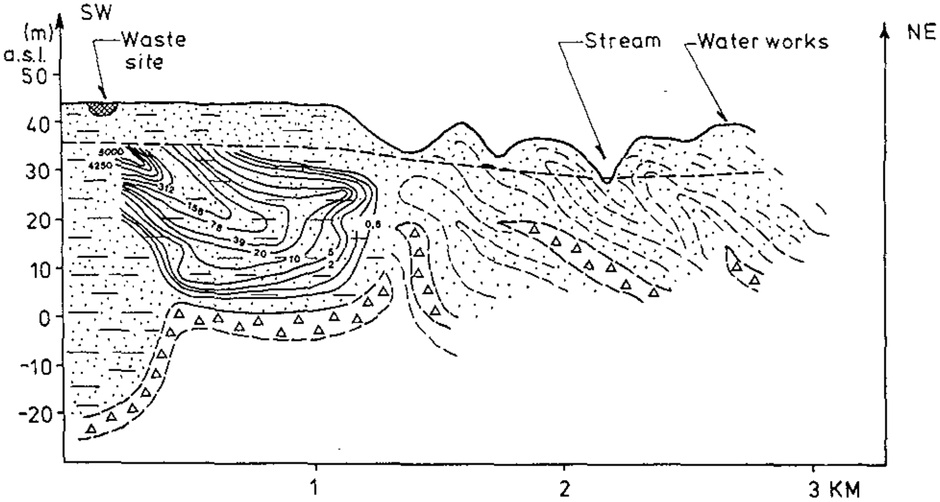

Figure 4 Vertical distribution along the plume axis of trichloroethylene + 1,1,1-trichloroethane in the groundwater reservoir

Retardation and degradation of solutes are implemented in the MOC model the latter as a first order degradation process. For the chlorinated compounds the literature suggests values for the retardation factor in the range from 1.5 to 5 and for the half-life period from 1/2 to 2 years. Calibrating the model to the observed measurements of concentrations applying the most probable distribution of the contamination source we obtained the following for the chlorinated compounds:

retardation factor	4,0
half-life period	3,5 year

After a simulation period of 23 years the MOC model gave a concentration distribution as shown in figure 3 - the simulated concentrations being close to the observed values.

The pilot water treatment plant consists of 2 aerators of the INKA type, sandfilters for the removal of solids, e.g. iron oxides, and 2 activated carbon filters. These units with the necessary pipes, pumps, electric installations etc. are built into a 20' container, giving a mobile plant which can be used as a pilot plant elsewhere or perform small scale clean-up at the rate of 5-10 m^3/h of water. A plan of the pilot plant is shown in figure 5.

In parallel to the INKA-aerators a stripping column of the Sultzer type has been evaluated.

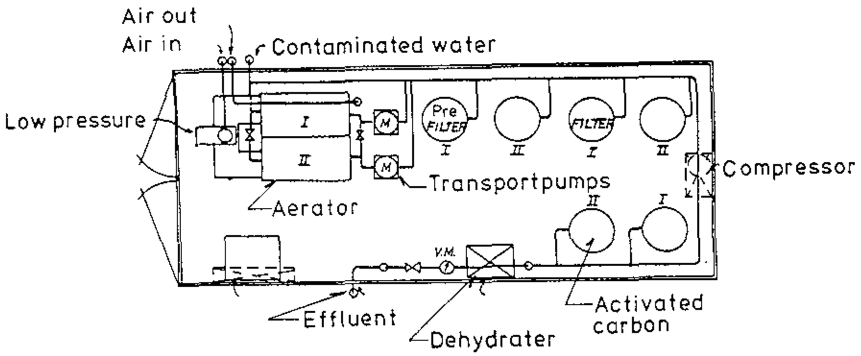

Figure 5 Mobile pilot water treatment plant for chlorinated solvents

RESULTS

Temporary collection site

The concentration of possible degradation products of 1,1,1-trichloroethane, measured relatively to the 1,1,1-trichloroethane concentrations, is shown in figure 6. The concentrations are shown in relative units as the absolute concentration of 1,1,1-trichloroethane varies randomly over a factor of 10.

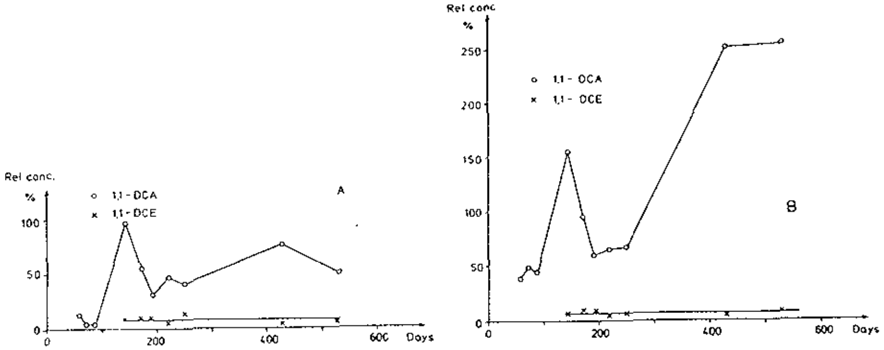

Figure 6 Concentrations of 1,1-dichloroethane (1,1-DCA) and 1,1-dichloroethylene (1,1-DCE) measured relatively to 1,1,1-trichloroethane in the aerobic (A) and the anaerobic (B) part of the collection site.

The concentration of 1,1-dichloroethylene is constant within analytical variability over the whole period. For 1,1-dichloroethane the variation is too large to allow any definite conclusions but a tendency to increased concentrations with time is apparent. Both substances are, however, known impurities in commercial 1,1,1-trichloroethane.

Results for the possible degradation of trichloroethylene is shown in figure 7.

Again there is no evidence for the formation of 1,1-dichloroethylene, while the results from the anaerobic part of the site give a clear indication for a formation of cis 1,2-dichloroethylene. Furthermore very small amounts of trans 1,2-dichloroethylene and vinylchloride (less than 1% relative concentration) are detected at the two last sampling dates. In the aerobic part of the site the concentration of cis 1,2-dichloroethylene also increases slightly, but the trans isomer and vinylchloride are not detected.

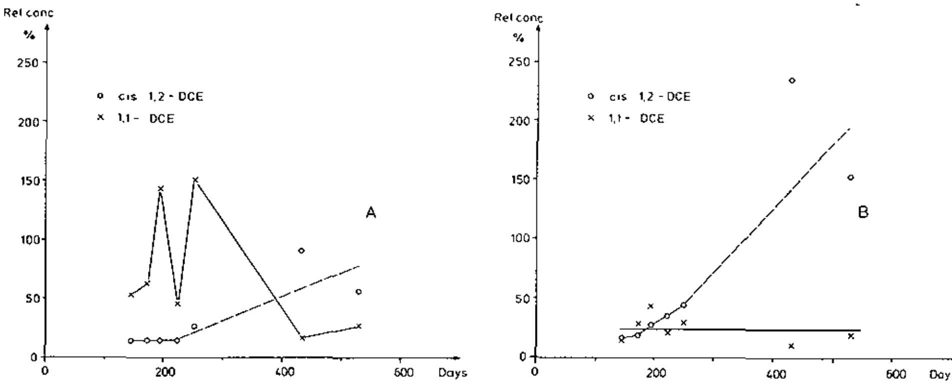

Figure 7 Concentration of cis 1,2-dichloroethylene (cis 1,2-DCE) and 1,1-dichlorethylene (1,1-DCE) measured relatively to trichloroethylene in the aerobic (A) and the anaerobic (B) part of the collection site.

These findings are in agreement with results from laboratory experiments (e.g. Vogel & McCarty, 1985) which show that cis 1,2-dichloroethylene is the dominant degradation product of trichloroethylene. Results from the litterature also show, that degradation preferably takes place under anaerobic conditions.

Ground water clean-up

Applying the USGS-MOC model the clean-up pumping yield has been optimized. Existing investigation wells are used as pumping wells assuming a characteristic yield of 110.000 m^3/year from each. The purification system is assumed to treat 3 x 110.000 m^3/year of contaminated water. By determining the rate of decrease of the **total mass** of the pollutants for several configurations of pumping the following results were obtained:

- pumping from the central part of the plume has the largest effect on the clean-up process
- the configuration of pumps is most important in the first phase (~ 2 years). After this period the concentration drops to a level where the pumps are equally efficient
- the clean-up process can be more efficient by pumping from partially penetrated wells close to the contamination source
- the clean-up will last from 10 to 20 years

In figure 8 the effect of pumping on the total mass of chlorinated solvents in the groundwater is shown. The mass drops from approximately 5000 kg to 50 kg after a 12 year period.

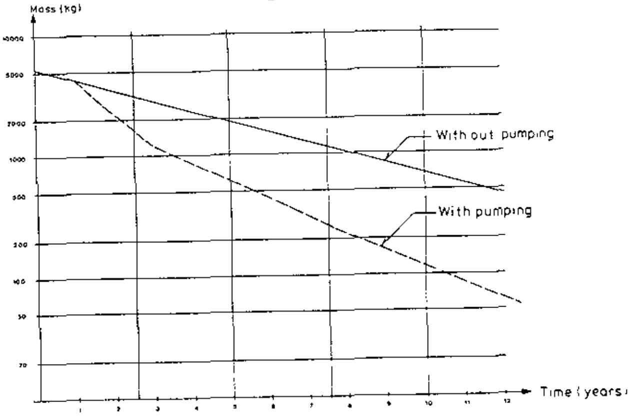

Figure 8 Effect of pumping on the total mass of chlorinated solvents compared to no pumping

Two types of aerators and activated carbon were evaluated in the pilot water treatment plant. The purification found for the two types of aerators is shown in table 3.

	Purification
INKA aerator	69 - 87%
Sultzer column	62 - 82%

Table 3 Purification of 1,1,1-trichloroethane and trichloroethylene by aeration

The results for the INKA-aerator are valid for air to water ratioes larger than 50. The two stripping systems give very similar results under these conditions, but the air consumption per m^3 of purified water is much larger for the INKA-aerator than for the Sultzer column. The INKA-aerator is, however, of a very simple construction requiring little attention, and thus this type of stripping system has been favored for the full scale plant.

Purification by activated carbon has also been tested, and the leaching curves for 1,1,1-trichloroethane and trichloroethylene in input concentrations of approximately 50 µg/l are shown in figure 9.

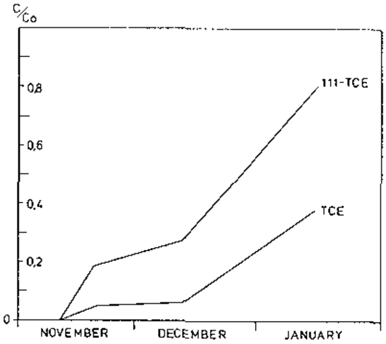

Figure 9 Leaching curves for 1,1,1-trichloroethane (1,1,1-TCA) and trichloroethylene (TCE) on activated carbon

The feed to the activated carbon columns contained, apart from the chlorinated solvents, ca. 400 µg/l of organophosphorous compounds and organic matter corresponding to a permanganate number of ca. 8 mg/l.

The leaching from the activated carbon columns starts quite soon. This may be due to organic matter in the feed water and probably also to a slight hydraulic overload of the columns during the experiments. The activated carbon has, however, a very good purification ability also at low feed concentrations and can, where needed, constitute a final cleaning step for chlorinated solvents.

The rationale for the use of activated carbon in the full scale plant is, however, the purification for organophosphorous compounds, which cannot be stripped from the water.

CONCLUSIONS

The results of the degradation experiments for soil in the temporary collection site have shown, that a degradation of chlorinated solvents is possible under anaerobic conditions. The process requires time, but very little attention.

Purification of water contaminated by chlorinated solvents can be performed using a low cost mobile aeration system built into a container. The system uses energy for the aeration, but like the soil degradation system it requires relatively little attention.

Optimizing the clean-up pumping rates and locations is possible applying a numerical flow and solute transport model with chemical reactions implemented to the site. With a total yield of 3 x 110.000 m^3/year the clean-up will last 10 to 20 years and the total mass of chlorinated solvents in the groundwater will decrease from 5000 kg to 50 kg during this period.

ACKNOWLEDGEMENT

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REMEDIATION OF HAZARDOUS WASTE CONTAMINATED LAND AT COMMERCIAL AND SMALL INDUSTRIAL SITES

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SUMMARY

Efforts to control the release of hazardous chemicals to the environment are increasingly being directed at small commercial and industrial facilities. Hazardous waste contaminated land is being discovered at or adjacent to a growing number of these facilities. While the associated human health or environmental threats are often not sufficient for the site to reach national priority status, they can be locally or regionally significant sources of pollution. Remediation of contamination at these sites can be complex and costly. Effective utilization of available investigation and cleanup technologies in a phased approach is necessary for a successful remediation effort. This paper presents a discussion of remediation concepts and methods followed by a case history describing investigation and cleanup activities at an equipment manufacturer in the central USA.

INTRODUCTION

The image brought to mind when one mentions the phrase, "hazardous waste site", is often one comprised of a huge chemical plant located in a heavy industrial center within a large metropolitan area. One is not surprised that such an apparently offensive facility has hazardous waste contaminated land on its property nor that it may cost a million dollars to cleanup the problems created.

In contrast to this image however, are the growing numbers of commercial and small industrial facilities where improper past or current waste management practices have resulted in hazardous waste being discharged to the land. It is surprising to many that extremely small quantities of certain chemicals can contaminate extensive land and water areas. For example, 1 L of certain organic solvents can contaminate nearly a million m^3 of water such that it is no longer fit for drinking. Because of this, many apparently innocuous commercial and small industrial properties have been confronted with costly remediation projects.

This paper presents a brief discussion of general remediation concepts and methods potentially appropriate for commercial and small industrial sites. A case history is given to illustrate the application of selected techniques at a small industrial site in the USA.

REMEDIATION CONCEPTS AND METHODS

Facilities such as listed in Table 1 are among the growing list where hazardous waste contamination has been discovered and where remediation has occurred. This is due to the fact that even very small quantities of hazardous wastes (e.g. 20 L) improperly discarded can cause significant risks to human health and the environment. The types of waste releases are numerous. Examples include dumping of liquids and sludges onto the ground surface or into seepage pits or lagoons; burial of waste containing drums, leaking containers and tanks; above-ground stockpiles; and uncontrolled burning.

Investigation and cleanup of these relatively smaller sites can present unique problems to decision makers involved in their remediation (Table 2). Ultimately decisions have to be made with many conflicting constraints and an information base which is frequently inadequate. There may be serious constraints on the remediation work due to pending regulatory violations and litigation for damages. Often, a significant driving force is the real threat of company bankruptcy.

Table 1. Example commercial businesses and small industries which generate hazardous wastes.[1]

Facility Operations	Example Hazardous Waste Components
Clothing Dry Cleaning	Chlorinated solvents.
Gasoline Stations	Aromatic hydrocarbons.
Mechanical Repair Shops	Aromatic hydrocarbons, chlorinated solvents, alcohols, acids, metals.
Wood Preserving	Phenols, metals, arsenates.
Coating and Painting	Organic solvents, metals, acids.
Printing	Metals, acids, solvents.
Electroplating	Metals, acids.
Electronics Manufacturing	Chlorinated solvents.
Equipment Manufacturing	Aromatic hydrocarbons, chlorinated solvents.
Agricultural Cooperatives	Herbicides, pesticides, acids.
Junkyards,scrap dealers	Solvents, acids, metals.

[1] This list is for general information purposes only and is not meant to be all inclusive nor completely accurate on a site specific basis.

Table 2. Problems potentially unique to remediation of hazardous waste contaminated land at commercial businesses and small industries.

Problem Category	Comments
Limited company knowledge	Company owners and personnel may be totally uninformed about proper management of hazardous waste nor the severity of penalties for improper practices.
Limited resources	Often the businesses involved can not begin to finance a hazardous waste remediation effort. Bankruptcy potential can be great.
Susceptible drinking water supplies	Many facilities are in suburban and rural areas which rely on ground water for drinking water. There is often limited testing or treatment.
Limited contamination	Often the extent of contaminated soil or water is significant but limited such that many technically attractive remediation alternatives are not cost-effective.
Limited local technical capability	Technical capability of the local consultants and regulators involved in the work can be incomplete.
Local politics	Since many procedures and protocols are not explicitly regulated, local politics with local officials, regulatory personnel and agencies can impact the course of a remediation project.

Remediation Concepts

One of the most perplexing and difficult questions yet to be fully resolved when confronting hazardous waste contaminated land is, "How clean is clean?" [1]. Approaches to establishing cleanup criteria vary widely and there is much debate over the best approach (Table 3). Many favor approaches based on site specific risk assessment methodologies. These require relatively precise characterization of the site in question in terms of the properties and toxicity of the contaminants present, transport pathways and receptors. With this information and a decision as to the level of acceptable risk, cleanup criteria can be established. This approach is currently employed in remediation efforts at national priority sites in the USA.

A generic approach is one where there are quality standards which all sites have to meet unless rigorous analyses can demonstrate that higher residual contamination is acceptably safe. Generic standards have been widely used for drinking water and more recently for ground water resources, regardless of their use as drinking water. This approach is being used for contaminated land at many non-national priority sites in the USA.

Table 3. Strategies for developing cleanup criteria for hazardous waste contaminated land.

Cleanup Criteria Basis	*Description*
<u>Cleanup Criteria Not Explicitly Set:</u>	
Diminishing Returns	Remedial actions carried out until regulatory agency deemed that further work would suffer *from diminishing returns.*
Designated Cleanup	Agreements to conduct specific remediation activities, without designated cleanup levels.
Background	Remedial actions required to mitigate contamination to background levels for the site in question.
<u>Cleanup Criteria Explicitly Set:</u>	
Arbitrary (?)	Based on general considerations of technical feasibility of cleanup, current and likely future land uses, qualitative risks, and legal negotiations. Criteria of non-detectable concentrations may fit here.
Risk Assessment	Either generic or site-specific risk assessment methods were used to set cleanup criteria.
Regulation	Cleanup criteria established by existing laws and regulations, often not specifically established for hazardous waste contaminated land (e.g. drinking water criteria).

A generic approach has the distinct advantage of providing a priori information regarding generally acceptable contamination levels and the nature and extent of cleanup likely. This facilitates effective project planning and execution. Generic standards are also needed for treatment facilities implemented for cleaning contaminated soil and water.

Regardless of the cleanup criteria established, considerations remain regarding their implementation. This includes decisions regarding the location and number of environmental samples collected, appropriate sampling and analyses methods, and statistical procedures for data analysis and interpretation. Often implementation is complicated by health and safety protocols and the need for real-time data collection and analysis during field operations.

Remediation Methods

Once hazardous waste contaminated land has been discovered, remediation activities normally commence. These can include immediate removal operations but more normally include planned site investigation activities followed by cleanup operations. For work at hazardous waste contaminated sites, field activities must account for health and safety hazards to onsite workers. This can complicate and markedly reduce the efficiency of ordinary field work and dramatically increase project costs.

A wide variety of site investigation techniques exist for characterizing hazardous waste contaminated land (Table 4) [2-5]. Moreover there is continuing research and development, especially in the area of remote sensing techniques and real-time data acquisition methods.

It is standard practice to employ individual techniques in a phased approach. It is essential that the work is well conceived and carefully executed due to health and safety considerations, litigation potential and other factors. The phased approach helps ensure that costs are contained while the knowledge and understanding of the site increases as necessary and appropriate. While appropriate techniques and phasing are highly site specific, many techniques are routinely employed as summarized in Table 4.

At different points in the investigation, the site characterization data is assessed in light of cleanup criteria, if known (see Table 3). Decisions are made regarding the need for and suitable methods of site cleanup. A wide variety of technologies have been attempted for remediation of hazardous waste contaminated land (Table 5) [6-9]. In addition there are refinements to existing technologies and new technologies continue to be developed.

Table 4. Example techniques employed in investigations of hazardous waste contaminated land.[1]

Investigation Phase		Methods
Initial Assessment	o	Industry operations records review.
	o	Personal interviews.
	o	Regulatory records review.
	o	Natural resource databases review.
	o	Site walkover and reconnaissance.
	o	Air quality screening.
Field Investigations	o	Historical air photography review.
	o	Historical topography and land use.
	o	Air quality monitoring.
	o	Topographic mapping.
	o	Surface geophysical surveys.
	o	Soil vapor surveys.
	o	Soil borings and testpits.
	o	Hydrogeologic testing.
	o	Soil contaminant sampling.
	o	Ground water monitoring.
	o	Surface water monitoring.
Detailed Site Assessment	o	Contaminant transport modeling and simulation.
	o	Risk assessment methodologies.

[1] The nature and extent of any particular method used on a specific project is highly dependent on the objectives and scope of the investigation and the characteristics of the site.

Unfortunately, while there are a diversity of technologies available, many are poorly suited to smaller sites due to their complexity and/or cost. Techniques have often been selected to expedite the cleanup effort while limiting operation and maintenance complexity, permitting requirements and costs. Buried wastes and grossly contaminated soils have often been transported for disposal at a offsite landfill. Soils contaminated with relatively low concentrations of organic compounds have been treated onsite via volatilization and/or biodegradation. For inorganics some form of solidification has been used. Ground water treatment has occurred via containment and pump and treatment using conventional water treatment processes (e.g. extended aeration, chemical precipitation, air-stripping, carbon adsorption).

Table 5. Example technologies which have been used for cleanup of hazardous waste contaminated land.

	Common Media	
Method	Soil	Ground water
Insitu Methods		
Passive	Yes	Yes
Volatilization	Yes	Yes
Enhanced Biodegradation	Yes	Yes
Stimulated Leaching	Yes	No
Vitrification	Yes	No
Containment	Yes	Yes
Non-Insitu Methods		
Land Treatment	Yes	Yes
Physical-Chemical Treatment	No	Yes
Biological Treatment	No	Yes
Thermal Treatment	Yes	No
Asphalt Batching	Yes	No
Solidification	Yes	No
Landfilling	Yes	No
Excavation	Yes	No
Pumping	No	Yes

[1] The suitability of any specific cleanup method is highly dependent on the objectives and scope of the cleanup and the characteristics of the site.

REMEDIATION CASE HISTORY

In response to an anonymous complaint the state pollution control agency conducted a surprise inspection of a small industry to investigate suspected hazardous waste management violations. Subsequent allegations led to a comprehensive investigation and cleanup project. This was conducted in a phased manner beginning in early 1987. The methods and results of this work are highlighted below.

Initial Assessment

An the onset, an assessment was conducted to elucidate industry operations and waste management practices, identify site natural resource features, assess health and safety concerns and determine if a remedial investigation was necessary and appropriate.

The industry in question was a small equipment manufacturing plant located in the north-central USA. The 40-yr old plant was located on a 20-acre (8.1 hectare) site which included offices, production facilities, a service garage, warehouses, gravel and asphalt driveways and grassy areas (Fig.1). The plant site was surrounded by residential development and agricultural fields.

Review of plant operations data indicated that hazardous chemicals were used now and/or in the past in painting and wood preserving operations, metal machining and fabrication, and in equipment and vehicle maintenance. Potential hazardous substances included heavy metals, organic solvents and phenolic compounds.

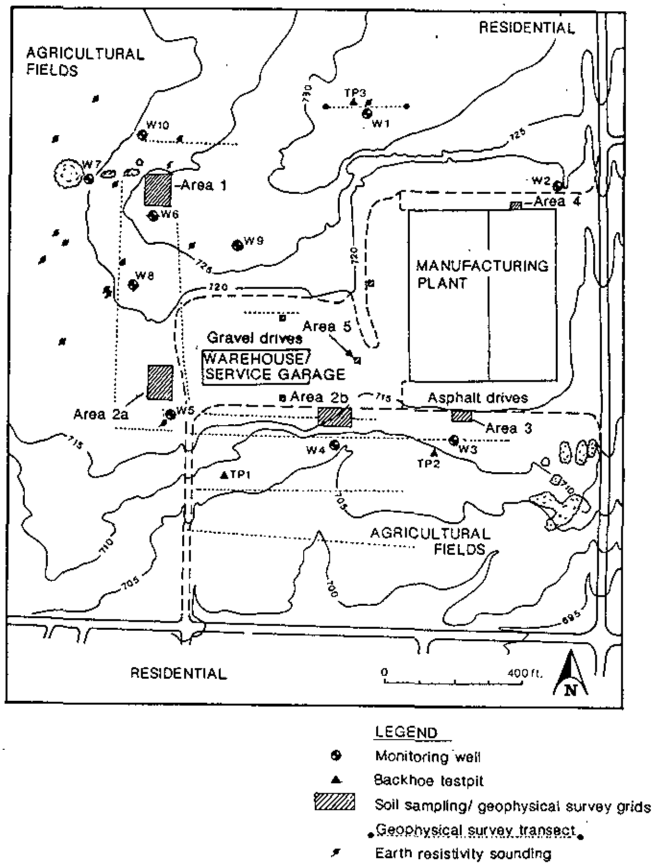

Figure 1. Topographic site plan of an equipment manufacturing plant with suspected hazardous waste contaminated land.

Interviews with plant personnel revealed five areas on the plant grounds which may have been impacted by past waste disposal practices (Fig.1). Area 1 included the site of a seepage pit, allegedly used for disposal of painting wastes until it was abandoned and filled with soil material during the early 1980's. Area 2 included two subareas where drums of paint sludges were allegedly buried some years earlier. Area 3 included an asphalt drive and grassy area upon which spent solvents may have been dispersed. Area 4 included an asphalt covered area where wood was preserved in dip tanks containing pentachlorophenol chemicals. Area 5 included gravel covered service drives where waste oils/fluids allegedly were dispersed.

Resource information indicated the site was characterized by sandy loam surface soils and till overlying dolomite bedrock at depths of 3 to 8 m. The local ground water table was reported to occur in the bedrock with horizontal flow toward wetland areas ca. 800 m away. Preliminary review of historical aerial photography and topographic data revealed modifications to the surface topography in the two areas where hazardous wastes were allegedly buried (Areas 1 and 2).

The acute health and safety risks associated with the alleged waste disposal activities at the plant were judged to be low for several reasons. The hazardous chemicals thought to be used were not acutely toxic and there were no signs of imminent release of chemicals or chemical waste. While drinking water in the area was derived from relatively shallow ground water, analyses of private wells in proximity to the plant revealed no contamination.

After assimilating the available information and conducting preliminary site inspections, a field investigation was deemed necessary and appropriate. It was also mandated by the state pollution control agency.

Phase 1 Field Investigation

The objectives of the Phase 1 Investigation were to characterize in detail, the nature resource and contamination characteristics at the site. The investigation consisted of topographic mapping, surface geophysical surveys, backhoe testpits, near-surface soil sampling, and ground water monitoring. Strict health and safety protocols (Level C/D) were followed during conduct of the work. This work was conducted during summer 1987 yielding the results described below.

The plant grounds occupied the crest and sideslope of a subtle ridge with land slopes varying from 3 to 30% (Fig.1). There was little surface water drainage onto the site. Observation and sampling in backhoe testpits revealed the surface soils to have pH values of 7.0 to 8.0 and organic matter contents of 1 to 2% (by weight).

To assess the potential presence of buried wastes, surface geophysical surveys were made [10]. Electromagnetic (EM) conductivity measurements were made with a Geonics EM-31 Terrain Conductivity Unit while magnetometry (MG) measurements were made with a Schonstedt GA-52B Fluxgate Gradiometer. EM conductivity data were first collected along continuous linear profiles (1400 m). Then based on the geophysical anomalies observed in concert with the initial assessment results, EM and MG measurements were made within rectangular systematic grids (1.5 - 3 m node spacings) established over the "hot spots". The data collected were analyzed using spatial statistical procedures (kriguing) and 3-D surface and isopleth maps were prepared.

The results of the geophysical work dispelled concern over several areas rumored to contain buried wastes, but confirmed three suspected waste burial areas (Areas 1, 2a and 2b). EM conductivity anomalies were distinct and readily apparent. Compared to background levels of 5 to 14 mmho/m, EM responses in the burial sites were 25 to 70 mmho/m, indicating the likely presence of buried waste materials (Fig.2). The MG data were less distinct, but generally consistent with the EM data.

Soil samples were collected from four areas alleged to have been impacted by surface releases of waste materials (Areas 1, 3, 4, 5, Fig.1), In three of the areas, systematic grids were established with node points at 1.5 to 3 m spacings. Over 100 samples were collected from depths of 0.1 to 1.1 m and screened in the field for volatile organic compounds using a photoionization detector (Photovac TIP) and headspace techniques. Based on the field screening, sensory observations and areal coverage considerations, 30

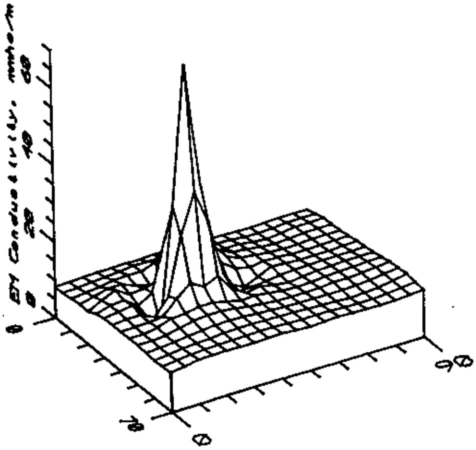

Figure 2. 3-dimensional map of EM conductivity in an area where there were allegedly buried drums of painting wastes (Area 2a).

samples were selected for detailed laboratory analyses. These samples were analyzed for basic soil properties and waste-derived contaminants, including purgeable organic compounds (34), phenolics (11) and heavy metals (8). The results of this sampling revealed generally low contamination by organic solvents and negligible concentrations of extractable heavy metals (Table 5).

Seven ground water monitoring wells were installed at six locations in close proximity to the alleged waste impacted areas. At each location a water table observation well was installed, while at one downgradient location (W4) there was also a deeper well (Fig.1). Head levels were measured on four dates while ground water quality analyses were made on two.

Ground water elevation data revealed the water table aquifer to vary in depth across the grounds from approximately 1.5 to 7.9 m. The water table was in glacial till and Paleozoic dolomite and appeared to flow to the south/southwest at a hydraulic gradient of 0.48%. Based on a single well nest, there appeared to be hydraulic discontinuities with depth.

Ground water quality analyses revealed significant, localized contamination in the water table well located at the abandoned seepage pit (W6, Area 1) (Table 6). In particular, the toxic organic compound, methylene chloride was measured as high as 40200 ug/L or ppb. This is substantially higher than the 150 ug/L state ground water and federal drinking water standard for that compound. There were also elevated levels of other less noxious contaminants.

Table 5. Selected results of soil sampling for organic contaminants during Phase 1 Field Investigations.[1]

Substance	Concentration Range	% of Samples in Which Detected	Areas in Which Detected
Toluene	20 - 2250 ppb	25 of 30	1,3,4,5
Chlorobenzene	15 - 81 ppb	9 of 30	1,3,-,5
Ethylbenzene	20 - - ppb	1 of 30	1,-,-,-
Methylene Chloride	16 - - ppb	1 of 30	-,-,4,-
Phenol	8 - 13 ppm	2 of 23	1,-,-,-

[1] Refer to Fig.1 for sampling area locations.

Table 6. Selected results of ground water quality sampling during Phase 1 Field Investigations.

Parameter	Units	Detect. Limit	Seepage Pit Well		Highest Value In Other Wells[1]
			Jul87	Aug87	
pH	-	-	7.9	8.3	7.2
Dissolved Solids	mg/L	20.0	1420	3020	844
Alkalinity	mgCaCO$_3$/L	20.0	1180	2350	784
Sodium	mg/L	0.03	472	1130	68
Chlorides	mg/L	1.0	55	74	57
Silver	ug/L	5.0	*	13	7
Arsenic	ug/L	5.0	28	*	11
Cadmium	ug/L	4.0	*	*	*
Chromium	ug/L	9.0	*	11	17
Mercury	ug/l	0.2	*	*	*
Nonpurgeable Organic Carbon	mg/L	0.1	89	335	8
Methylene Chloride	ug/L	5.0	7690	40200	*
Phenol	ug/L	2.0	66	194	*
2,4-Dimethylphenol	ug/L	3.0	*	8	*
Head	m	0.01	215.6	215.4	215.9
	feet	0.1	707.2	706.6	708.2

[1] Based on measurements for each of 5 wells (W1-W5) in July and August 1987.

Phase 1 Cleanup

Based on the findings of the Phase 1 Investigation, cleanup operations were initiated. The objectives were to excavate, examine and exhume any buried waste materials in the areas encompassing an alleged abandoned seepage pit (Area 1) and two alleged waste burial sites (Areas 2a and 2b). Again, health and safety protocols (Level C) were followed during conduct of the work which occurred during fall 1987.

Prior to cleanup operations cleanup criteria were discussed with the state pollution control agency. Firm criteria were not explicitly set but tentative goals were discussed. Any material classified as hazardous waste per federal hazardous waste regulations criteria had to be handled as such. The approach of diminishing returns (see Table 3) was generally to be employed for cleanup of contaminated soils, but residual levels of total volatile organic chemicals (VOCs) were to be 1 mg/kg or less.

Each of the areas was excavated with a track-mounted excavator beginning at the peak of the geophysical anomaly identified during the Phase 1 work (e.g. Fig.2). Excavation was continued outward until the buried waste materials were exposed and removed. This work revealed that there were indeed buried waste materials at each of the three suspect areas (Table 7). In all three areas the boundaries of the waste materials coincided with those of the geophysical anomaly. The buried wastes contained appreciable levels of organic solvents and heavy metals. The exhumed materials, ca. 70 m^3 of chemical sludge and 100 drums of paint waste, were transported in bulk to a licensed hazardous waste landfill. Several drums of semi-solids were stabilized with cement and disposed in a similar manner.

Table 7. Summary characteristics of buried wastes exhumed during Phase 1 Cleanup operations.

	Buried Waste Characteristics	
Area Characteristics	Description	Contaminants Detected
Area 1 - Abandoned and buried seepage pit for painting wastes:		
Sloping, unobstructed, sandy loam soils/till to bedrock at 4.6 m depth. Groundwater in bedrock at 6.1 m depth.	60 cm sludge blanket at 1.5-2.1 m depth in 93 sq.m area. 70 m^3 sludge were exhumed.	Lead = 5030 ppm Chromium = 970 ppm Meth.Chlor.=4900 ppm Ethylbenzene=200 ppm Toluene = 9 ppm Xylenes = 830 ppm
Area 2a - Burial site for crushed drums of waste and debris:		
Level, barren, sandy loam soils/till to bedrock at 3.4 m depth. Groundwater at 4.6 m depth. Natural depression that was filled	Random mixture of crushed drums of dry solids and other debris at 0.3 -1.8 m depth in 104 sq.m area. 50 208-L metal drums were exhumed.	Lead = 3500 ppm Chromium = 1700 ppm Ethylbenzene=3900ppm Toluene = 70 ppm Xylenes = 14300 ppm
Area 2b - Burial site for intact drums of waste:		
Level, barren, sandy loam soils/till to bedrock and ground water at 7.6 m depth. Filled area near edge of embankment.	Orderly side-stacking of intact drums of solids and semisolids at 1.8 - 3.0 m depth in 74 sq.m area. 50 208-L metal drums were exhumed.	Lead = 280 ppm Chromium = 60 ppm Ethylbenzene=170 ppm Xylenes = 560 ppm

The soil materials surrounding the buried wastes exhibited varying contaminant levels as outlined in Table 8. For the two buried drum sites, the soils were lightly contaminated. Considering the nature and concentrations of residual contaminants and the current and likely future land uses, the excavation spoil was used to backfill these two excavations and the sites were considered adequately clean.

At the abandoned seepage pit, the deeper overburden soils were more heavily contaminated as were the sub-pit soil and till materials. Due to the high residual levels of methylene chloride, lead and chromium, further remediation actions were contemplated. Consideration is being given to soil remediation by land treatment of the spoil materials (ca. 75 m^3) and insitu volatilization of the subpit materials. Long-term passive remediation would then be relied upon after closure of the pit.

Table 8. *Selected results of soil contaminant sampling during Phase* 1 Cleanup operations.

Disposal Area	Soil Contaminant Concentrations Total VOCs[1] (ppm)	 Total Lead (ppm)	 Total Chromium (ppm)
Area 1 - Abandoned and buried seepage pit for painting wastes:			
Overburden soil	2 (- %)[2]	17	9
Deep Overburden	32 (14%)	419	102
Sludge	5900 (82%)	5030	970
Subsoil (-15 cm)	115 (40%)	21	12
Subtill (-60 cm)	53 (26%)	15	10
Area 2a - Burial site for crushed drums of waste and debris:			
Overburden soil	1.2 (*%)	11	10
Deep Overburden	0.3 (*%)	404	79
Drummed waste	18200. (*%)	3504	1710
Subsoil (-15 cm)	5.4 (*%)	10	12
Subsoil (-75 cm)	*[3] (*%)	2	5
Area 2b - Burial site for intact drums of waste:			
Overburden soil	0.1 (*%)	4	5
Drummed waste	732. (*%)	284	59
Subsoil (-60 cm)	0.02(*%)	20	10

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[1] Total volatile organic compounds (VOCs) = sum of analyses for 34 different purgeable organic compounds by purge and trap and gas chromatographic techniques.

[2] Number in parenthesis indicates percent of indicated Total VOCs that were halogenated.

[3] "*" indicates analyzed for but not detected.

Phase 2 Field Investigation

In concert with the cleanup operations, a Phase 2 Field Investigation was initiated to further characterize the apparent ground water contamination in the vicinity of the abandoned seepage pit and assess the necessity and options for ground water cleanup. This work included surface geophysical studies, a soil pore gas survey and additional ground water monitoring. Remote sensing techniques were employed to assist in siting additional ground water monitoring wells. All of this work was conducted during late 1987 and through 1988.

Electromagnetic measurements were made with a Geonics EM-34 Terrain Conductivity Unit (10 and 20 m coil spacings) at several locations around the seepage pit. However, due to local soil and geologic properties, these measurements did not identify any conductivity anomalies. The earth resistivity survey appeared more insightful. Vertical and horizontal earth resistivity soundings were made at 17 locations around the seepage pit with an ABEM 300 Terrameter with a Bison Boss Offset Sounding System and a Wenner array with A-spacings of 0.5 to 32 m. The results of this work revealed subtle conductivity anomalies in the vicinity of the abandoned seepage pit. The apparent anomalies were localized to the pit area and ca. 60 m to the south and southeast of it.

Soil gas samples were collected at 7 locations around the perimeter of the seepage pit within ca. 100 m of it. The samples were collected with a 2 cm diam. probe driven to 1.5 to 2 m depth. After evacuation of the tube, gas samples were collected and immediately analyzed for methylene chloride and other solvent compounds using field portable gas chromatography equipment. Ground water samples were also collected from monitoring well W6 adjacent to the seepage pit. In the ground water sample, the methylene chloride concentration was ca. 25000 ug/L. However minimal methylene chloride or other solvents were detected in the headspace in the well or in the soil pore gas. This was attributed the volatilization properties of methylene chloride and more importantly, the dense glacial till barrier above the ground water. This provided a barrier to the upward transport of any gases potentially volatilizing from the ground water zone.

Based on the results of the Phase 1 ground water monitoring and the Phase 2 geophysical survey, five additional ground water monitoring wells were installed at five locations around the perimeter of the abandoned seepage pit (Fig.1, W6a-W10). Ground water elevation data were collected in December 1987 and June 1988 from the five newly installed wells and from seven existing wells.

The results derived from installation and monitoring of these new wells revealed that within ca. 75 m of the seepage pit, the water table aquifer occurred at a depth of approximately 4 to 6 m while at the pit perimeter it was ca. 3.4 m below the pit bottom. In December, 1987, the horizontal hydraulic gradient in the ground water was measured at 0.40% to the south/southwest (Fig. 3). Interestingly, the groundwater flow was roughly perpendicular to the apparent slope of the bedrock surface (Fig. 4).

The ground water elevation measured in the deep piezometer near the seepage pit was consistent with that measured at the other onsite piezometer. Comparison of the ground water elevations in the water table wells versus the deep piezometers at both locations suggested there was an aquitard at some zone between approximately 5 and 13 m depth which created a hydraulic discontinuity. However, no strong lithologic evidence of this feature was observed in the boring process.

The Phase 2 ground water quality sampling yielded some perplexing results. Compared to the ground water composition observed beneath the seepage pit in July and August, 1987, contaminant concentrations were consistently and substantially lower in December 1987 and June 1988 (Table 9). Despite a relatively constant head level, this trend was consistent for several pollutants (Fig.5).

In the four water table wells around the pit (ca. 55 to 85 m away from the pit), the ground water composition observed in December 1987, was similar to that observed in the other onsite wells away from the abandoned seepage pit. However, during June 1988, contaminants in the perimeter wells increased somewhat (Table 10). In particular, methylene chloride was detected in all three wells potentially downgradient from the pit. Ground water extracted from the deeper well near the seepage pit generally exhibited high quality. However, methylene chloride was detected on both sampling occasions, but at very low levels (< 1.7 ppb).

The reasons for these ground water quality observations are currently unknown. Due to estimated contaminant transport properties in the seepage pit area, it is unlikely that the ground water quality trends could be attributed to cleanup operations (i.e. removal of the contaminant-laden sludge blanket). It is more likely that these results are due to ground water flow and contaminant transport dynamics in the vicinity of the seepage pit. Related to this, during drilling of well W6 near the seepage pit, substantial voids were encountered in the dolomite above the water table. Pockets of concentrated leachate from the pit may release chemicals intermittently based on water table fluctuations or other factors.

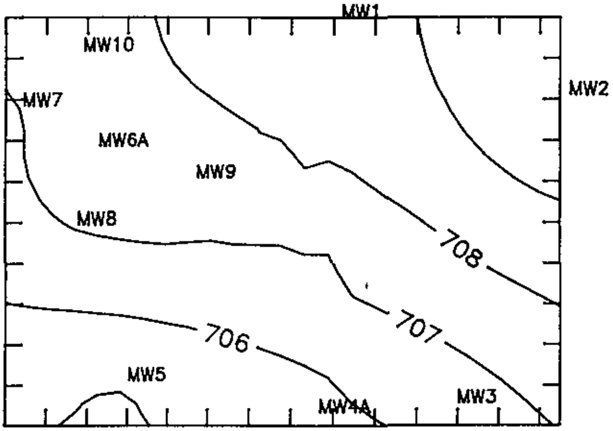

Figure 3. Ground water table surface topography as estimated by point measurements and geostatistical modeling methods (based on head levels in 10 water table observation wells in December 1987; contour interval = 1 foot).

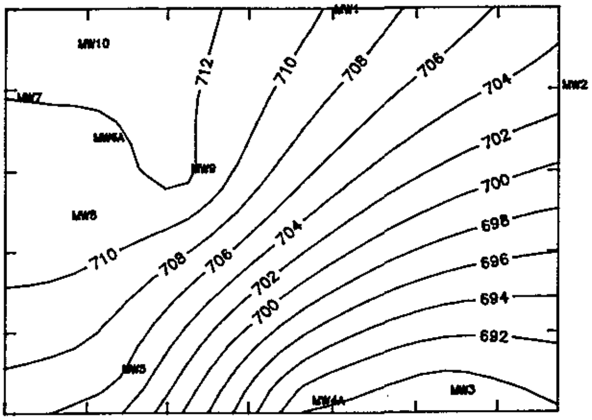

Figure 4. Bedrock surface topography as estimated by point measurements and geostatistical modeling methods (based on lithology in 10 auger borings; contour interval = 2 feet).

Table 9. Selected results of ground water quality measurements near an abandoned seepage pit allegedly used for disposal of painting wastes.[1]

	Ground Water Concentrations				
Sampling Date	Total Dissolved Solids	Nonpurge. Organic Carbon	Methylene Chloride	Total Xylenes	Phenol
	mg/L	mg/L	ug/L	ug/L	ug/L
July 1987	1420	89	7020	*[2]	66
August 1987	3020	335	40200	*	202
December 1987	1360	56	440	48	*
June 1988	604	9	1131	*	*

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[1] The monitoring well (W6) was a water table observation well within ca. 16 m of the seepage pit center (Fig.1).

[2] "*" indicates analyzed for but not detected.

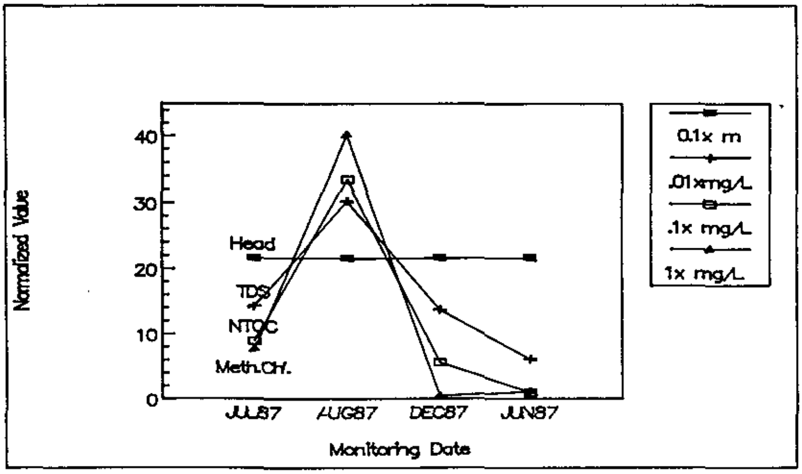

Figure 5. Normalized temporal variation in ground water characteristics adjacent to an abandoned seepage pit.

Status of Remediation Efforts

Considerable progress toward remediation has been made relatively quickly and at moderate cost. However, further work remains. There is soil remediation and closure of the abandoned seepage pit.

Precise characterization of the ground water flow and contaminant transport properties in the vicinity of the abandoned seepage pit may be impossible due to shallow aquitards and fractured media. However, further characterization work is in progress. The expanded ground water database will be evaluated to determine the necessity for further remediation. If needed, cleanup design and implementation would follow. Finally, regulatory and legal issues remain to be fully resolved.

The impacts to industry and the local community have been significant. To date, the industry has incurred over $300,000 in direct costs associated with the investigation and cleanup. There have also been indirect costs. Management and corporate counsel efforts have been diverted from productive business. There has been anxiety and stress by industry owners and there has been some diminished employee morale due in part, to suspended payments to the company pension plan. Public images may have been tarnished. Finally, there are uncertain future liabilities and costs for associated damages.

Table 10. Selected results of ground water quality sampling in the vicinity of an abandoned seepage pit allegedly used for disposal of painting wastes.[1]

	Total Dissolved Solids		Methylene Chloride	
Monitoring Well	December 1987	June 1988	December 1987	June 1988
	-------- mg/L --------		------- ug/L ---------	
W7	492	696	*[2]	80.4
W8	536	589	*	4.8
W9	364	466	*	6.1
W10	424	450	*	*
W6	1360	604	443.	1131.
W6b	352	441	1.3	1.7

[1] All monitoring wells were water observation wells within ca. 85 m of the seepage pit center, except well W6b which was a piezometer next to well W6 and positioned ca. 6 m deeper than W6 (see Fig.1).

[2] an "*" indicates the compound was not detected at a detection limit of 1 ug/L.

CONCLUSIONS

The problems associated with remediation of hazardous waste contaminated land can be complex and costly even on commercial and small industrial properties. In fact, there are problems potentially unique to the remediation of these relatively smaller sites. A wide variety of remediation technologies exist which should be employed based on project and site specific considerations. The remediation investigation and cleanup should be well-conceived and executed in a phased approach. Generally, the consequences, both direct and indirect, of even a small remediation effort can be devastating. Therefore, efforts should be expended to alert and educate key persons to prevent future problems.

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Strategy, forecasting and policy

On investigation and redevelopment of contaminated sites.

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History.

In connection with a national survey in Denmark carried out in 1980-82 it was estimated that 500 out of 3.000 registered dumps/industrial sites were contaminated with chemical wastes.

On the basis of these 500 chemical dumps, the Danish Parliament passed the chemical waste sites act in 1983.
At the same time 400 mill. Dkr. were appropriated. It was expected that the sum of money was enough to cover the costs of investigations and remedial actions and that the authorities over a 10 years period would gain control over the chemical waste sites.

However, in the period from the implementation of the law and till now a lot of new chemical waste sites have been recorded. In May 1988 the total amount of these sites was 1599.

The considerable increase in the number of recorded sites is due to the fact that the original survey in 1980-82 primarily was concerned with the large-volume disposal sites.

To day the authorities know that industrial sites where chemicals were handled and deposited can cause considerable groundwater contamination.

Another aspect is towndevelopment on former industrial sites. Thus in addition to the groundwater aspect there may be problems concerning the human exposure to hazardous chemicals from the soil - inside or outside the houses.

In 1982 the industrial sites only constituted 5% of the recorded sites. This percentage was in 1987 estimated to be as high as 65%.

The increase of recorded sites is expected to be just as drastic in the coming years, because the mapping of the industrial sites is not finished. Mapping this type of contaminated sites is a difficult and demanding job. It implies knowledge of the relevant type of production, the plants and their use and handling of chemicals.

Until all of the contaminated sites have been recorded we expect to run into acute problems like contaminated wells or contaminated soil when for example excavation is started in connection with construction work.

The experience gained from 5 years of administration of the waste sites law is that 10 years and 400 mill. Dkr. are far from enough to solve the problems.

The total amount of sites has increased drastically, and the cost of investigations and remedial actions exceeds the level which was assumed from the beginning.

Aim.

In spite of the size and complexity of the problems, the National Agency of Environmental Protection (NAEP) in Denmark, still intends to handle this type of contamination in a way that exposure of the environment and human beings to hazardous chemicals is minimized. Particularly it is hoped to save our groundwater-resources and surface recipients (biotopes) and to avoid that human beings are exposed by using the sites.

It is very important for Denmark to solve the problems of contaminated soil thoroughly, because we primarily use groundwater for drinking water and because of a demand for town-development on industrial sites.

Strategy.

Our strategy is first of all to avoid new local depositions of hazardous chemicals. Important means to obtain this is the compulsory national collecting system of chemical waste. In this way burying or discharging the hazardous chemicals should be avoided to a great extent.

The law of environmental protection ensures that newer plants have to be approved so that they will not give rise to soilcontamination. NAEP also finds development of "cleaner technology" important to solve these problems.

The increase in the recorded sites is therefore mainly expected to come from the old industrial sites.

The principles of recording.

To get a more precise knowledge of the extent of the industrial waste sites problem the NAEP has thoroughly analysed use and handling of chemicals in relevant plants.

On the basis of this work some types of plants are known to give rise to soilcontamination. Former or actual locations of such plants the NAEP finds relevant to record.

Priority.

To solve the most threatening problems immediately the authorities put the efforts in order of priority. The ranking is based on a model where the elements partly are:

- the relevant chemical components on the sites
- the actual concentrations in the soil and groundwater
- the behaviour and dispersion of such chemicals in soil, air and water
- the knowledge of toxicology of the chemicals

The model also includes the localisation of the actual sites related to the groundwaterresources - the geology and hydrogeology - and the actual use of the sites (residential areas, industrial areas, or recreational areas).

Strategy of investigations.

When a contaminated site is going to be investigated, often there are different sorts of risks that must be elucidated, for instance

- risk of contamination of groundwater and recipients
- risk of pollution of air, outdoors and indoors
- risk of pollution of accessible soil and crops, grown at the site.

It is not allways possible to recognize these risks before the practical part of the investigation is started, and the extent and type of contamination may turn out to be different from what was expected. Therefore, it is necessary to divide the investigations into phases, so that after a phase of orientation one may make a goal-directed limited investigation of a certain part of the site or a limited number of hazardous substances.

The investigations of contaminated sites usually are divided into:

- a phase of orientation
- one or more phases of investigations
- a phase of remedial actions
- a phase of running and controlling certain remedial actions

The purpose of the phase of orientation is roughly to map the source of contamination, the strength of it and its chemical composition. The phase of orientation is furthermore intended to describe the need of succeeding investigations and give concrete suggestions.

The purpose of the next phase is to carry out and report the investigations that are necessary to determine the extent and the consequences of the contamination, and to plan remedial actions if necessary. Monitoring by controlling the remedial action may mean that you have to turn to more phases og investigations and the following phases.

It is decisive for the applicability of the investigations that the authorities and their consultants have a procedure of securing the quality of the work. This concerns especially the decisions about the strategy of investigations, the methods of taking samples, the analyses at the laboratory and the reporting of the results.

<u>The role of risk assesments</u>

Risk of contamination of groundwater may only determine part of the strategy for investigations of a contaminated site. In planning the *investigations* attention should also be drawn to possible risks of exposure of human beings and their environment by direct contact to the hazardous substances in the soil or to dust, vapour or crops containing the substances.

The NAEP recommends the local authorities, that areas of the contaminated soil to which sensitive users may be exposed are investigated in most detail. By accomodating the remedial actions or the use of the site to the results of the investigations and the risk assesments, risks to users are minimized.

Experiences from cases of contaminated sites have shown that the public usually will not accept involuntarily to be exposed to hazardous chemical substances, for example on a residential site. This is the case also when the estimated risk that can be related to hazardous substances in the soil is small. The general attitude at the authorities is actually that the exposure to hazardous substances from contaminated sites only is allowed to be minimal in view of the fact that it is only part of the total exposure of humans from various sources. When determining demands to remedial actions at the contaminated sites, it is also stressed that the solution ought to be valid for a long time into the future.

Remedial actions

To protect groundwater from contamination, remedial actions are established according to the localisation of the hazardous substances in relation to the groundwater of interest and to the vulnerability of the groundwaterresources.

A conventional method is for example excavation of "hot spots" combined with hydrological steering/pumping of the contaminated water.

Newer methods more and more gain footing for example on-site treatment of soil and groundwater by wet-oxidative or microbiological processes. Furthermore one uses on-site treatment methods, where soil and groundwater are decontaminated for example by recirculation of water rinsed by passing through a coalfilter.

If the hazardous waste sites are localised in environments where there is potential exposure of human beings and their surroundings - apart from groundwater resources of interest - the remedial actions will primarily be directed against the contaminated soil. The possibilities of exposure may in this connection be minimized by excavation, in-situ or on-site treatment of the contaminated soil or, as mentioned before, by choosing a less vulnerable use of the site. For instance one may, instead of making residential area with gardens, cover the surface soil and put a factory at a part of the site where there may be insignificant risk of indoor problems from the soil.

Research

To optimize the remedial actions and cut down the expenditures the Danish authorities, research institutes and consultant firms try to build up know-how. Many resources are engaged in research within this field.

The researchers are primarily concerned with developing alternative methods of soil treatment, as the conventional thermic methods are disproportionately expensive. At this moment experiments in laboratories and in pilot field trials are promising especially within extraction and microbiological processes.

In the field of treatment of groundwater Denmark is quite advanced regarding research within <u>in-situ</u> and on-site methods reaching from wet-oxidative processes to methods using reverse osmosis.

Furthermore Denmark gives research within risk assessment some priority - including assessments of risks to the health of users of the sites - in order to optimize remedial actions according to critical exposure routes.

<u>Barriers</u>

Practising the strategies gives, as may be expected, some problems. The greatest barrier the authorities are facing is that solution of the cases of contaminated sites cannot keep pace with the number of recorded and expected problem sites.

It is quite a job in itself to map the extent of the problem, as recording the contaminated sites demands knowledge of the industries through different periods in each town. Each recorded case then demands individual evaluation as to planning the investigation, analyse the results and take the proper action. So the limited number of people engaged in this sort of work partly determines the rate of the solutions. At the same time we are working under relatively *narrow economical conditions*. It should be mentioned here that the main principle is that the polluter pays, but in the many cases where it is not possible to get hold of payment from a former polluter, the state and the local authorities share the expenses in connection with sites of high priority.

Some sites are given low priority because their present use does not cause problems with groundwater and health concerns. Private interests in altering the use of these sites mean however, that quite a number of sites are so-called "voluntarily cleared". Here the costs of all investigation and remedial actions are covered by the owner.

Working with contamination of soil and groundwater from hazardous chemical substances implies a thorough knowledge of many different subject fields. So it is necessary with a well functioning collaboration among people with knowledge within hydrology, geology, engineering, chemistry, biology, toxicology and public medicine.

The migration and behavior of the chemical substances in soil and groundwater are relatively unexplored subjects, and many years are going to pass before one will be able to describe the different aspects of it thoroughly.

This is part of the reason why ranking of the efforts spent on clean-up and reclamation of contaminated soil, rests on a rather narrow base. Sometimes we are surprised by cases, where groundwater of interest already is contaminated or cases of people already having been exposed to hazardous chemicals as a consequence of using the site.

Despite the fact that we must face barriers of lack of knowledge, the work within the field has, during the latest years, given valuable experience. The building up of knowledge internationally and within our own country escalates during these years and there is a hope that we, by combining the implicated subject fields, get a solid base to manage and solve the problems efficiently. But building up the needed know-how demands wide economic frames, and in the end it is a question whether a limited budget will force us to leave the Danish principle that clean drinking water and soil that is safe to live on are basic demands.

From NAEP we will express the hope that we get possibilities to maintain the principle mentioned, and that the problems may be solved at a rate that let a minimum of irreversible damages happen.

SUPERFUND

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INTRODUCTION

In response to public concern over poor past disposal practices of hazardous wastes and the thousands of uncontrolled waste sites throughout the United States, in December 1980 Congress enacted the Comprehensive Environmental Response, Compensation and Liability Act, PL 96-510 (1), commonly known as CERCLA, or Superfund. This act was a result of the wide publicity that sites such as Love Canal, Valley of the Drums, and Stringfellow had received that made the public aware that uncontrolled hazardous waste sites were either cousing harm to the public health and environment or were a potential threat in the future.

The superfund Act was enacted in part by the recognition that the Recourse Conservation and Recovery Act, commonly referred to as RCRA or Solid Waste Act, provided a regulatory program for active hazardous waste facilities, but not inactive sites. Superfund provided a Federal response to uncontrolled releases of hazardous substances from a vessel or from any onshore or offshore facility. A key feature of the Act was the establishment of a trust fund to support the cleanup activities. This fund received $ 1,6 billion over a 5-year period. The revenue was supported 87.5 percent by taxes on petroleum and chemical feedstock, and 12.5 percent from a Treasury appropriation. This authority ran out in December 1985. After almost a year without new authority, extensive amendments known as the Superfund Amendments and Reauthorization Act, PL 99-499 (2), called SARA, was adopted in October 1986. This Act established a new fund of $ 8.5 billion over a 5-year period beginning January 1, 1987. The principal sources of revenue will be a tax on petroleum, 42 listed chemical, a broad-based corporate environmental tax, and appropriations from general revenues.

Although the U.S. Environmental Protection Agensy (USEPA) has been severely criticized by Congress for lack of action in the cleanup of hazardous waste sites, its record of activities during the first 5 years has been extensive (3). The USEPA developed an inventory of 25,194 hazardous waste sites throughout the United States. A preliminary assessment of 20,023 was made. A total of 1,477 sites was scored using the Hazardous Ranking system to evaluate the health and environmental threat posed by the sites. The USEPA placed 703 sites on The National Priority List (NPL) and 185 more sites ware proposed for listing. Since this report, even more sites have been proposed. Some form of response action had been taken at 1,174 sites and emergency response actions, often called removal actions, had been commenced at 808 sites. Of these sites, 611 had been financed by the government, 150 by the potentially responsible parties (PRP), and 47 by both ths USEPA and the PRP´s. Remedial Investigations and Feasibility Studies (RI/FS) were begun at 468 sites and long-term remedial actions were completed at 14 NPL sites. The value of PRP response actions was estimated at $ 818 million.

OVERVIEW OF CERCLA AND SARA

As specified in its name, CERCLA is a very comprehensive law, directing the USEPA, other Federal agencies, and the states on how to collect and manage the "fund", conduct cleanups, encourage PRP´s to conduct their own cleanups, and recover the costs of Federally supported actions from the PRP´s. The 1980 legislation established a fairly straightforward structure to obtain these goals. The 1986 SARA amendments did not fundamentally alter the structure, but added far more detail than was present in the original law and added some additinal requirements and provisions. In fact, the length of the amendments exceeded that of the original law.

The detailed new provisions speak of such issues as "how clean is clean", by defining cleanup standards and a preference for permanent treatment and on-site remedies. Deadlines with specific numbers of actions that must be taken for evaluating sites, placing them on the NPL, and undertaking response actions are included. Health effects study requirements, applicable to every site on the NPL, were added. A new title was added on emergency planning and community right-to-know in response to concerns raised over catastrophic releases of toxic chemicals, e.g., the Bhopal, India, disaster where 2,500 people were killed.

RELEASES, HAZARDOUS SUBSTANCES, REPORTABLE QUANTITIES

Section 103 of Superfund specifies that responsible parties must report to the Federal Govenment´s National Response Center (800/424-8802) any spill or other release of a hazardous substance to the environment in a reportable quantity. Congress defines a "release" as any spill leaking, pumping, pouring, emitting, emptying, discharging, injection, escaping, leaching, dumping, or disposing into the environment, including the abandonment or discharging of barrels, containers, and other closed receptables containing any hazardous substance, pollutant, or contaminant (2).

What is a "hazardous substance"? Superfund defines a "hazardous substance" as (1) any substance designated as a hazardous pollutant under Section 311 of the Clean Air Act; (2) any toxic pollutant listed under Section 307(a) of the Clean Water Act; (3) any hazardous air pollutant listed under Section 112 of the Clean Air Act; (4) any hazardous waste under RCRA; (5) any imminently hazardous chemical substance or mixture with regard to which the USEPA Administrator has taken action under Section 7 of the Toxic Substances Control Act (TSCA); and (6) any substance designated as hazardous under Section 192 of Superfund. When all of these lists were combined into a "list of lists", 717 hazardous substances were identified (4).

The term "hazardous substance" does not include petroleum, natural gas, or synthetic gas or pipeline quality. In addition, certain solid wastes are suspended from regulation. These include solid waste from the extraction, beneficiation, and processing of ores and minerals, cement kiln dust, and waste generated from the combustion of coal or other fossil fuels.

Superfund also requires the USEPA to establish regulations on "reprotable quantities". This requirement establishes the quantity for each hazardous substance, the release of which shall be reported. "Reportable quantities" have been promulgated for 442 hazardous substances based on scientific criteria relating to the risk posed by a release of that substance (4). The USEPA is evaluating the remaining 275 hazardous substances for potential carcinogenicity and chronic toxicity. Until the USEPA sets a reportable quantity for these substances, the quantity previously established under Section 311 of the Clean Water Act, or, if none exists, the reportable quantity is 1 pound.

The reportable quantities start at 1 pound for many substances and range up to 5,000 pounds for substances such as chloroform, carbon tetrachloride, and hydrocloric acid. For mixtures and solutions there is a "release" when a component hazardous substance is in the mixture or solution in its reportable quantity.

NATIONAL CONTINGENCY PLAN (NCP)

The cornerstone of the removal and remedial *program of* Superfund is the NCP. Section 105 of Superfund (1, 2) specifies the content of this plan. The plan establishes the procedures and standards for responding to releases of hazardous substances, pollutants, and contaminants. In addition, it contains the National Priorities List (NPL). This list contains those sites that present the greatest danger to public health and welfare or the environment, and thus are eligible for cleanup action under Superfund. The NCP includes matters such as:

1. Methods for discovering and investigating facilities at which hazardous substances have been disposed or otherwise came to be located;

2. Methods for evaluating, including analysis of relative cost, and remedying any releases or threats of releases from facilities which pose substantial danger to the public health or the environment;

3. Methods and criteria for determining the appropriate extent of removal, remedy, and other measures;

4. Appropriate roles and responsibilities for the Federal, state, and local governments, and for interstate and nongovernmental entities in effectuating the plan;

5. A method for and assignment of responsibility for reporting the existence of such facilities which may be located on federally-owned or controlled properties; and

6. Criteria for determining priorities among releases or threatened releases throughout the United States (Hazardous Ranking System, (HRS)).

The NCP was substantially revised and last reissued in November 1985 (5). SARA requires several changes in the NCP and a revised plan will be issued by the USEPA in 1987 and 1988. There are seven phases for determining the appropriate response to a release of hazardous substances:

1. Discovery and notification
2. Preliminary assessment and evaluation
3. Removal actions
4. Site evaluation and NPL determination
5. Community relations
6. Remedial action
7. Documentation and cost recovery

DISCOVERY AND NOTIFICATION

The response to a hazardous release starts when the USEPA is informed that a release has occurred or is imminent. This notification can come from several sources including: a notification to the National Response Center; notification by the owners or operators of a facility; or through investigations by state and Federal government authorities. Over 25,000 sites have been identified in this manner (3).

PRELIMINARY ASSESSMENT AND EVALUATION

A notification of a release will result in the USEPA undertaking a preliminary assessment to evaluate the need for a "response action". The assessment includes an evaluation of the magnitude of the hazard, the identification of any party or parties who are ready, willing, and able to undertake a response action, and an evaluation of whether immediate removal is necessary. Readily available information and data are reviewed at this stage. Available records, photographs, ect., are collected and personal interviews are made. A perimeter inspection of the site to assess the potential for a release may also be taken.

The preliminary assessment will be terminated when it is determined that there has been no release; the release does not involve a hazardous substance, pollutant, or contaminant;

the amount released does not require a federal response; a responsible party is providing an appropriate response; or the assessment is completed.

If the assessment indicates that a "removal action" is necessary, (Removal Actions are discussed in the next section), then the USEPA is required to identify responsible parties and have them conduct a removal action or conduct the removal themselves. If the assessment indicates a "remedial action" is necessary, then the procedures specified in the NCP for a remedial action is undertaken.

REMOVAL ACTIONS

Removal actions are undertaken when the USEPA determines that there is a threat to public health or welfare, or the environment. This determination is made after reviewing the following factors:

1. Is there an actual or potential exposure by hazardous substances or contaminants to nearby populations, animals, or food chains?

2. Is there an actual or potential contamination of drinking water supplies or sensitive ecosystems?

3. Is there hazardous substances or pollutant, or contaminants in drums, barrels, tanks, or other bulk storage containers that may pose a threat of release?

4. Is there a threat of fire or explosion?

5. Are there high levels of hazardous substances and soils that may migrate?

6. Are there weather conditions that may cause hazardous substances to migrate?

The purpose of a removal action is to abate, minimize, stabilize, mitigate, or eliminate a public health threat. In essence, a removal action is usually done to abate the immediate threat of a hazardous waste to the public health or environment. It usually does not lead to a complete cleanup of a site. The complete cleanup follows the procedures described in the NCP.

Removal actions may include as simple an action as the installation of a security fence and warning signs to prevent access of humans and animals to the release or destruction of the on-site, or the removal of the waste to approved RCRA facilities for treatment or disposal. Removal actions have included activities such as installation of drainage controls to prevent runoff of contaminated waters; capping of contaminated soils and sludges, segregation and storage of drummed wastes, removal of highly contaminated soils, sludges, or drums; providing alternative water supplies; and incineration or treatment of waste on-site.

SARA places a limit on removal actions of 2 million dollars and/or the lepse of 12 months from the date of the initial removal, unless it is found that circumstances warrant continued action. Because of the "emergency" nature of removal actions, they are not required to meet other federal, state, or local requirements such as obtaining permits. However, removal actions are required to the greatest extent practicable to meet or exceed applicable or relevant and appropriate federal public health and environmental requirements.

If the removal action does not sufficiently abate the threat, an orderly transmission from a removal to a remedial action occurs. By mid-1986, the USEPA had commenced removal actions at 808 sites.

SITE EVALUATION AND NATIONAL PRIORITY LIST

Those sites that the USEPA decides warrant further evaluation after completion of a preliminary assessment undergo a site inspection (SI) (6). SI's generally include on-site and off-site observations and sampling to identify the presence of hazardous substances and to determine whether off-site migration has occurred. SI should complete the collection of data so that an evaluation of the threat to public health and the environment can be determined. This model is called the Hazard Ranking System (HRS).

The HRS is one of the most significant processes under Superfund because the decision to list or not list a particular site on the National Priority List is based on the HRS ranking. An HRS listing of a site assures evaluation of that site to determine the exact nature and extent of the risk that it presents and the effective remediation alternative, i.e., the Remedial Investigation/Feasibility Study (RI/FS) process. Listing a site also means the federal fund money could be spent on the site and that potentially responsible parties (PRP) will be identified for cleanup cost or action.

The HRS (7) evaluates relative risk among hazardous waste sites. Risk is a composite measure of the probsbility and magnitude of adverse effects, and is specified in terms of what the risks are and who is at risk. In the first case, the concern is the "risk of" a release, exposure, health effect, and/or ecological damage. The second case is the "risk to" an individual, a population group, and an environmental resource. A hazardous waste site usually poses a variety of risks to a variety of targets, and the HRS is intended to rank sites in terms of their overall relative risk to humans and the environment. a uniform application of the ranking system enables EPA to make a technical judgment regarding the potential hazards presented by a facility relative to other facilities. It does not address the feasibility, desirability, or degree of cleanup required.

The HRS assigns three scores to a hazardous facility:

- S_M reflects the potential for harm to humans or the environment from migration of a hazardous substance away from the facility by routes involving groundwater, surface water, and air. It is a composite of separate scores for each of the three routes.

- S_{FE} reflects the potential for harm from substances that can explode or cause fires.

- S_{DC} refelcts the potential for harm from direct contact with hazardous substances at the facility (i.e., no migration need be involved).

The score for each hazard mode (migration, fire and explosion, and direct contact) or route is obtained by considering a set of factors that characterize the potential of the facility to cause harm (Table 1). Each factor is assigned a nummerical value (on a scale of 0 to 3, 5, or 8) according to prescribed guidelines. This value is then multiplied by a weighting factor yielding the category are added; then the total scores for each factor category are miltiplied together to develop a score for groundwater, surface water, air, fire and explosion, and direct contact.

In computing S_{FE}, S_{DC}, or an individual migration route score, the product of individual category scores is devided by the maximum possible score, and the resulting ratio is multiplied by 100. The last step puts the score on a scale of 0 to 100.

S_M is a composite of the scores for the three possible migration routes:

$$S_M = \frac{1}{1.73} \qquad S^2 + S^2 + S^2$$

where:

S_{gw} = groundwater route score
S_{sw} = surface water route score
S_a = air route score

The effect of this means of combining the route scores is to emphasize the primary (highest scoring) route in aggregating route scores while giving some additional consideration to the secondary or tertiary routes if they score high. The factor 1/1.73 is used simply for the purpose of reducing S_M scores to a 100-point scale.

The HRS does not quantify the probability of harm from a facility or the magnitude of the harm that could result, although the factors have been selected in order to approximate both elements of risk. It is a procedure for ranking facilities in terms of the potential threat they pose by describing:

- the manner in which the hazardous substances are contained,
- the route by which they would be released,
- the characteristics and amount of the harmful substances, and
- the likely targets.

The multiplicative combination of factor category scores is an approximation of the more rigorous approaching which one would express the hazard posed by a facility as the product of the probability of a harmful occurrence and the magnitude of the potential damage.

The ranking of facilities nationally for remedial action will be based primarily on S_M. S_{FE} and S_{DC}may be used to identify facilities requiring emergency attention. Details on the calculation of HRS scores are presented in Reference 7.

The Superfund Amendments (SARA) required EPA within 18 months after the enactment of the Act to amend the HRS to assure, to the maximum extent feasible, that the hazard ranking system accurately assess the relative degree of risk to human health and the environment posed by sites and facilities subject to review. The amended HRS should become effective within 24 months of the enactment or no later than October 1988.

As of June 1986 (8), 703 sites had been placed on the National Priority List and another 185 sites had been proposed (Figure 1). Table 2 presents a list of the NPL sites by state/territory. New Jersey, Michigan, New York, Pennsylvania, and California lead the list with 97, 66, 65, 65 and 61 respectively. Figure 2 and Table 3 show the types of activities at each of the 888 sites and Figure 3 shows the observed contamination.

REMEDIAL INVESTIGATION/FEASIBILITY STUDY (RI/FS)

Once a site has been placed on the NPL, the U.S. EPA, a state, or the PRP´s will conduct a Remedial Investigation (RI) and prepare a Feasibility Study (FS) leading to a decision for a remedial action or "no-action". Remedial actions are responses to releases that preferably result in a permanent remedy. The RI/FS is used by EPA to determine the nature and extent of the threat and to evaluate and select a remedy.

The remedial investigation and the feasibility study are interdependent. The activities comprising these two projects are generally performed concurrently rather than sequentially.

The remedial investigation emphasizes data collection and site characterization, whereas the feasibility study emphasizes data analysis and decision-making.

The remedial investigation is conducted concurrently with the feasibility study. The remedial investigation is the data collection mechanism for the feasibility study effort; this relationship is discussed later in this chapter. The remedial investigation also supports remedial alternatives evaluation through bench and pilot studies. Figure 4 illustrates the remedial investigation process.

The initial activity in the remedial investigation is the scoping process. The Scoping effort includes the collection and evaluation of existing data, identification of remedial investigation objectives, and the identification of general response actions for the feasibility study. Data needs, preliminary plans, and investigation tasks are identified. The investigation scoping process may recur or be modified as more data are collected and site characterization becomes more complete.

The scoping process is critical to the development of a sampling plan and subsequent remedial investigation. This sampling plan describes the sampling studies to be conducted, including sample types, and sampling locations and frequency. Planning needs such as sampling operational plans, material, record-keeping, sampling team personnel needs, and sampling procedures are also developed or identified for the investigation.

Associated with the scoping and sampling plan efforts are a variety of support activities that require the preparation of specific plans or implementation of specific procedures to supplement the remedial investigation and documentation of data. Specific plans address data management procedures, including quality assurance/quality control programs; health and safety planning requirements, including development of an overall health and safety program and a site-specific health and safety plan; and may review institutional issues arising from Federal, state, and local regulations, policies, and guidelines that may affect the investigation.

Site characterization involves the collection and analysis of the data needed for the various types of assessments that are part of the investigation. Because site data and understanding vary, a multilevel approach to data collection is recommended. Each level differs in the scope of the activities. The three levels of data collection and site characterization efforts are:

- <u>Level I - Problem Identification and Scoping.</u> Existing site information is collected and evaluated to define the problem(s) at the site, piblic and environmental threats, and site features contributing to the problem(s). This assessment is conducted for all sites and provides the basis for immediate mitigation actions for defining investigation needs in levels II and III. The data collected at this level are used in identifying and analyzing remedial technologies.

- Level II - Problem Quantification. Specific site data are collected through sampling and field studies to characterize site problems and their dimensions more fully. Sufficient data should be collected to identify contaminations of concern, to verify actual axposure pathways, and, in general, to characterize the site well enough to support, at a minimum, the screening of remedial technologies and alternatives.

- Level III - Problem Quantification and Detailed Investigation. If level II data are insufficient, additional data are collected for use in detailed analysis of remedial alternatives or in the selection of a costeffective alternative.

The remedial investigation does not require that all three levels be completed; the process may terminate at any level provided that sufficient data have been obtained. For some sites, a level I study may furnish enough data for response decisions, particularly if a site has been well-studied or the need for an immediate response is obvious. The investigation may end at level II if characterization data are sufficient to permit the selection of a response. Alternatively, where level I analysis are sufficient to support feasibility study decisions and a level II effort is not necessary, a level III study involving bench or pilot testing may be needed to select between alternatives or finalize a design. Thus, the investigation needs vary from site to site, and the levels of the remedial investigation must be appropriate to these needs.

Bench- or pilot-scale studies may be needed in the remedial investigation to obtain enough data to select a remedial alternative. The scope of bench and pilot studies in the remedial investigation specifically addresses waste treatability, scale-up of innovative technologies, technology application issues, and evaluation of specific alternatives. Bench and pilot studies may also be conducted during remedial alternative design or construction to more fully evaluate specific requirements of the selected alternative, however, these studies are outside the remedial investigation and feasibility study process. In general, bench-scale studies are appropriate for the remedial investigation stage while pilot-scale studies, if required, may be conducted during the final design.

Figure 5 depicts the concurrent activities associated with the remedial investigation (RI) and feasibility study (FS). The upper portion of the figure consists of two flow charts illustrating the sequential, interdependent events associated with the RI/FS process. The lower portion of the figure is a tabulation of the tasks identified in the Model Statement of Work for the RI/FS. This model Statement of Work sets forth the tasks that a contractor will perform in conducting a government-led

RI/FS. The lower portin of Figure 5 also identifies the chapters in the Remedial Investigation and Feasibility Study Guidance Documents (9, 10) that the U.S. EPA has produced.

The vertical lines on the chart indicate some of the plans, reports, or milestones recommended in the RI/FS guidance. These connectors and the listings below them illustrate the integration of the RI/FS process.

The feasibility study process is outlined in Figure 6. The first step of the feasibility study, defining the objectives of the action and broadly developing response actions, should be performed as a refinement to project scoping during the remedial investigation and should be summarized in the final remedial investigation report. There may be modification of this scoping as data are collected or as the general response actions are more fully developed during the feasibility study stage. The remedial alternatives developed at this point are general response actions which broadly define the nature of the response. In general, they should address whether source control measures designed to prevent or minimize migration of hazardous substances from the source and/or measures designed to mitigate the impact of contamination that has migrated into the environment are necessary, and what phasing of these measures may be necessary.

The next step in the process is the development of specific alternatives within the general response categories. First, technologies within the categories are screened for their technical applicability to the site. Technologies considered technically appropriate are then combined to form operable units that address one or more aspects of the identified site problems. These operable units may then be combined to form alternatives addressing the complete site. The alternatives are then screened on the basis of public health, environmental, and cost concerns.

The next five activities comprise the detailed analysis of alternatives, which is necessary to provide the decision-maker with information for selecting the alternative that is cost-effective. These activities include the engineering analysis of the alternatives in terms of constructability and reliability to ensure the implementability of the selected remedial action; an evaluation of institutional requirements such as compliance with other applicable or relevant and appropriate (ARA) Federal and State environmental statutes and community relations; an evaluation of public health protection requirements that must be met; an evaluation of environmental impacts of the action; and a detailed cost analysis.

A major issue in all remedial actions is "how clean is clean", that is, to what extent should a site be cleaned up.

Congress has provided a clear directive in SARA for the selection of a permanent action. Section 121 provides that "remedial actions in which treatment which permanently and significantly reduces the volume, toxicity, or mobility of the hazardous substances.....are to be preferred", and that off-site transport and disposal without treatment is the last favored method of completing remedial actions. This provision was to discourage untreated land disposal, or simply move waste from location to another without reducing the long-term threat. To underscore the fact that treatment is preferred over land disposal or containment, the U.S. EPA must review a site every 5 years to ensure that the final solution continues to be protective of human health and the environment of the remedial action involves leaving the hazardous substance at the site.

If hazardous substances (or pollutants or contaminants) will remain at the site, i.e., on-site remedies, the remedial action must result in a level or standard of control for the substance which at least attains the legally applicable standards, requirements, or criteria under federal environmental laws (including the Resource Conservation and Recovery Act, the Toxic Substances Control Act, the Safe Drinking Water Act, the Clean Air Act, and the Clean Water Act or state environmental or facility siting laws that are more stringent than federal standards. If land disposal is chosen as part of the remedial action for on-site remedies, any state standard that would effectively result in a statewide prohibition of the land disposal shall not apply except in certain narrowly-defined circumstances. No federal, state, or local permit is required for actions taken entirely on-site, if the remedial action is carried out in compliance with the cleanup standards.

If an off-site remedy is chosen, the haxardous substances must be transferred to a facility that is operating in compliance with a RCRA or other federal permit. The substances can only be sent to a land disposal facility if (a) the unit to which the substances will be sent is not releasing any hazardous wastes into the groundwater, surface water, or soil, and (b) any releases from other units at the facility are being controlled by a corrective action program.

There are six statutory exceptions to the new requirement that the remedial action meet the applicable or relevant environmental standards and criteria appropriate requirements (ARAR). These include cases where the remedial action is an operable unit and only a part of a larger action that will attain the appricable levels when completed; compliance would result in greater risk; compliance is technically impracticable; the remedial action selected will attain a standard of performance that is equivalent to the required standard; the state standard at issue has not been consistently applied within the state; and in the case of a

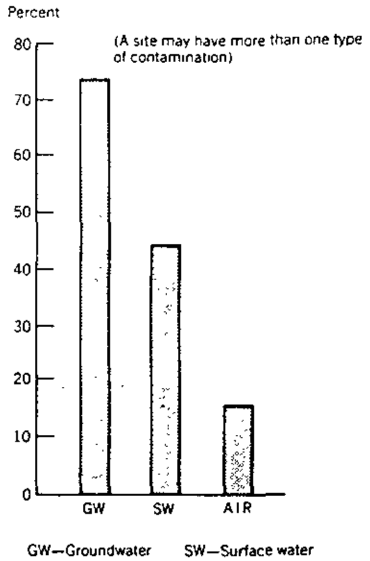

Figure 3 Observed contamination at 888 final and proposed NPL sites, June 1986.

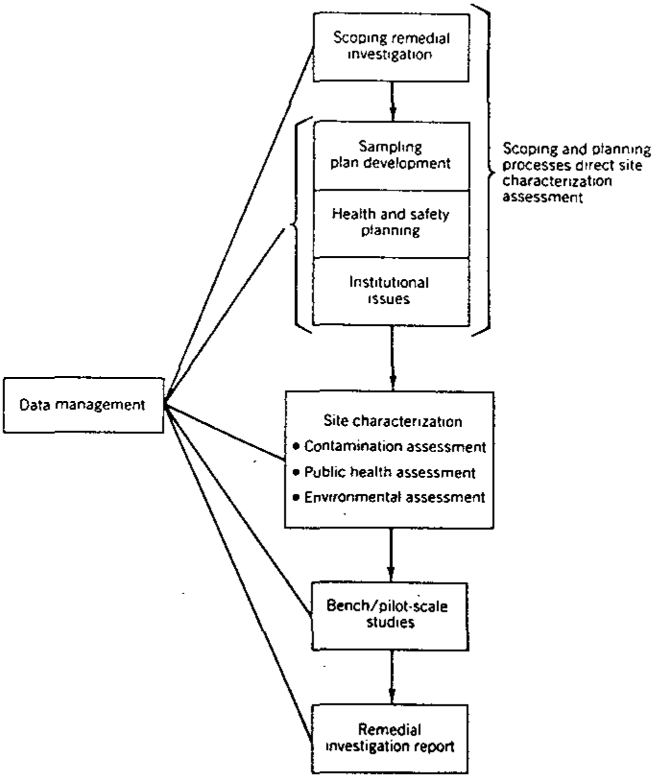

Figure 4. Remedial investigation process.

medial action is begun along with a response to significant comments, criticisms, and new data. The U.S. EPA is also required to publish an explanation of any part of the completed remedial action that differs from the final plan.

As of June 1986, 126 RODs or enforcement decision documents (EDD) had been signed.

REMEDIAL ACTIONS

By the end of 1986, the U.S. EPA had completed 14 remedial actions (RA) and another 103 had been initiated. SARA mandates that 175 new RAs be initiated during the first 3 years after enactment and an additinal 200 RAs be initiated during the fourth and fifth year after enactment.

Management techniques for hazardous wastes found at Superfund sites may be grouped into two broad categories--land disposal (either on-site or off-site) and use of technologies that permanently reduce or destroy the hazardous character of the waste. In general, land disposal was used more frequently than other management techniques in the first 5 years of Superfund. This is because, for many waste types, land disposal is typically less expensive than technologies which change the nature of the waste. As a result, hazardous waste management at Superfund sites has frequently relied on relatively inexpensive land disposal. This reliance has limited the development and use of typically more expensive options. The heavy reliance on land disposal of hazardous wastes is illustrated in Figure 7. This figure summarizes a review of remedial activities at 51 National Priority List (NPL) sites which revealed that on-site land disposal, including capping and on-site landfilling, was used at 52 percent of the sites and off-sites land disposal was used at 54 percent of the sites.

Concern over the long-term reliability of land-based disposal practices promted Congress to direct the Agency to give preference to providing treatment which permanently and significantly reduces the volume, toxicity, or mobility of the hazardous substance. Since technology is not readily available for many Superfund problems, SARA authorized a research, development, and demonstration program. The U.S. EPA established the Superfund Innovative Technology Evaluation (SITE) program to find better solutions to hazardous waste cleanup (11). As part of the SITE program, the U.E. SPA is planning to evaluate the most promising new technologies in several demonstration projects each year at Superfund sites across the nation. Testing may involved only one technology at a site or a set of them at various stages of the cleanup investigation. The U.S. EPA will choose the specific Superfund sites after a nationwide search that will match the effectiveness and compatibility of various technologies with specific wastes and conditions at the selected sites. The demonstrations will generate data on the cost and effetiveness of the technology.

TABLE 3. Types of Activities at 888 Final and Proposed NPL Sites in Order of Occurrence, June 1986

Surface impoundments
Commercial/industrial landfills
Containers/drums
Municipal landfills
Manufacturing other than chemical
Spills
Chemical processing/manufacturing
Leaking containers
Tanks, above-ground
Waste piles
Groundwater plumes
Tanks, below-ground
Wood preserving
Electroplating
Waste-oil processing
Ore processing/refining
Surface-water outfalls
Military ordnance
Solvent recovery
Open burning
Land farm/land treatment
Incinerators
Military testing/maintenace
Battery recycling
Surface mining sites
Underground injection
Drum recycling
Road oil
Sand and gravel pits
Sink holes
Subsurface mining sites
Explosive disposal/detonation
Laundry/dry cleaning
Tire storage/recycling

remedial action under Section 104 using the Fund, a remedial action that complies with the standard will not provide an appropriate balance between the need for protection of public health and the environment and the monies available in the Fund.

As of late 1986, the U.S. EPA had initiated 489 RI/FS´. SARA established a mandatory time schedule for the U.S. EPA to conduct RI/FS´ in the future. Within 3 years, 275 RI/FS´ must be initiated and within 5 years a total of 650 RI/FS´ must be initiated.

The RI/FS´ conducted to date have provided some valuable information on Superfund site characteristics. Table 4 lists the substances most frequently found at sites.

RECORD OF DECISION

Once the RI/FS process is complete, the U.S. EPA selects a response action. The Agency is required to establish an administrative record called Record of Decision, or "ROD", in support of the selection of a response action for the site. The ROD serves as a basis for judicial review of Agency action and must be available to the public. Public participation must take place in the development of the ROD. The U.S. EPA is required to publish a notice and analysis of a proposed plan and allow opportunity for comments and public meetings. It also requires the final plan to be published and made available to the public before a re-

TABLE 1. Of Comprehensive List to Rating Factors

Hazard Mode	Factor Category	Factors		
		Groundwater Route	Surface-Water Route	Air Route
Migration	Route characteristics	Depth to Aquifer of concern Net precipitation Permeability of unsaturated zone Physical state	Facility slope and intervening terrain One-year 24-hour rainfall Distance to nearest surface water Physical state	
	Containment	Containment	Containment	
	Waste characteristics	Toxicity/persistence Hazardous-waste quantity	Toxicity/persistence Hazardous-waste quantity	Reactivity/incompatibility Toxicity Hazardous-waste quantity
	Targets	Groundwater use Distance to nearest well/population served	Surface-water use Distance to sensitive environment Population served/distance to water intake downstream	Land use Population from within 4-mile radius Distance to sensitive environment
Fire and explosion	Containment	Containment		
	Waste characteristics	Direct evidence Ignitability Reactivity Incompatibility Hazardous-Waste Quantity		
	Targets	Distances to nearest population Distance to nearest building Distance to nearest sensitive environment Land use Population within two-mile radius Number of buildings within two-mile radius		
Direct contact	Observed incident	Observed incident		
	Accessibility	Accessibility of hazardous substances		
	Containment	Containment		
	Toxicity	Toxicity		
	Targets	Population within one-mile radius Distance to critical habitat		

TABLE 2. Final and Proposed NPL Sites per State/Territory (by Total Sites) June 1986

		Proposed*		
State/Territory	Final NPL	Nonfederal	Federal	Total
New Jersey	91	3	3	97
Michigan	56	10	0	66
New York	57	7	1	65
Pennsylvania	48	14	3	65
California	34	19	8	61
Florida	32	7	0	39
Minnesota	36	2	0	38
Ohio	27	3	0	30
Wisconsin	26	4	0	30
Indiana	23	5	0	28
Washington	19	3	6	28
Texas	21	3	2	26
Illinois	14	7	4	25
Massachusetts	21	0	0	21
Missouri	12	3	2	17
Colorado	12	1	2	15
Delaware	9	4	1	14
Iowa	6	7	0	13
New Hamsphire	12	1	0	13
Virginia	7	5	1	13
South Carolina	10	2	0	12
Alabama	8	0	2	10
Kentucky	9	1	0	10
Utah	3	4	3	10
Arizona	5	4	0	9
Montana	7	2	0	9
Arkansas	7	1	0	8
Maryland	6	0	2	8
North Carolina	6	2	0	8
Puerto Rico	8	0	0	8
Rhode Island	8	0	0	8
Tennessee	7	0	1	8
Connecticut	6	1	0	7
Kansas	6	1	0	7
Louisiana	5	1	1	7
Maine	5	1	1	7
Hawaii	0	6	0	6
West Virginia	5	1	0	6
Georgia	3	1	1	5
Nebraska	2	2	1	5
Oklahoma	4	0	1	5
Oregon	4	0	1	5
Idaho	4	0	0	4
New Mexico	4	0	0	4
Mississippi	2	0	0	2
Vermont	2	0	0	2
Guam	1	0	0	1
North Dakota	1	0	0	1
South Dakota	1	0	0	1
Wyoming	1	0	0	1
Alaska	0	0	0	0
American Samoa	0	0	0	0
Commonwealth of Marianas	0	0	0	0
District of Columbia	0	0	0	0
Nevada	0	0	0	0
Trust territories	0	0	0	0
Virgin Islands	0	0	0	0
	703	138	47	888

*Includes 45 proposed update no. 5 sites, 28 proposed update no. 4 sites, 24 proposed update no. 3 sites, 86 proposed update no. 2 sites, two proposed update no. 1 sites.

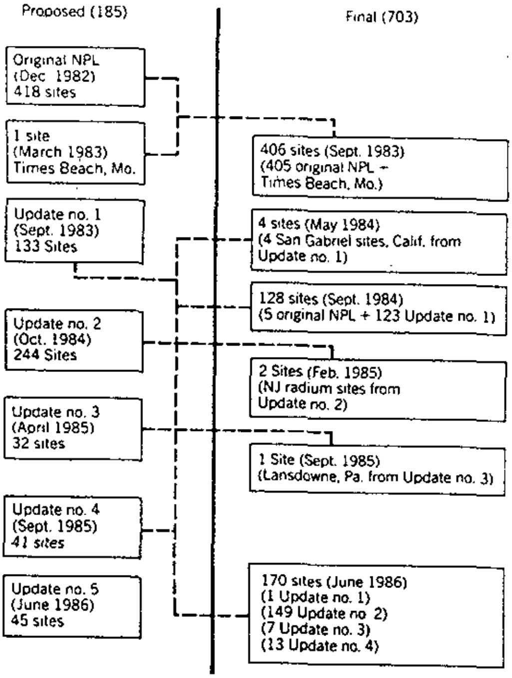

Figure 1. Development of NPL.

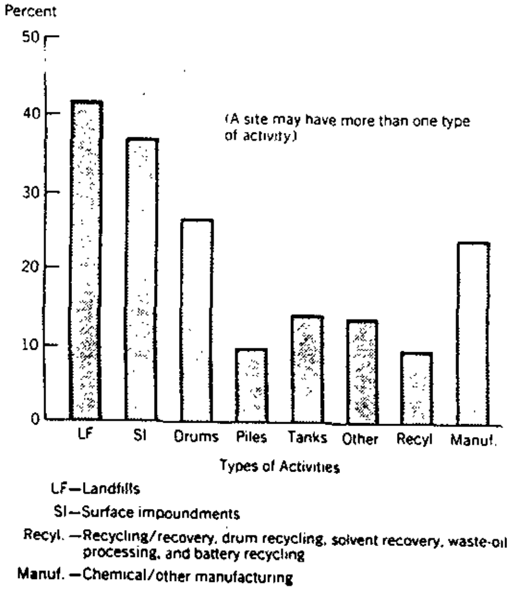

Figure 2. Types of activities at 888 final and proposed NPL sites, June 1986.

TABLE 4. Substances Most Frequently Found at Superfund Sites

Substance Name	Site Frequency
Trichloroethylene (TCE)	297
1,1,2-Trichloroethylene	297
Lead (Pb)	273
Toluene	235
Benzene	204
Chloroform	179
Polychlorinated biphenyls, NOS	159
1,1,2,2,-Tetrachloroethene	145
Zinc and compounds, NOS (Zn)	137
1,1,1-trichloroethane	137
Cadmium (Cd)	132
Arsenic (As)	127
Chromium and Compounds, NOS (Cr)	125
Phenol	124
Ethylbenzene	106
Xylene	106
1,2-Trans-dichloroethylene	104
Copper and compounds, NOS (Cu)	98
Methylene Chloride	93
Chromium	80
1,1-Dichloroethane	75
Mercury (Hg)	75
Cyanides (soluble salts), NOS	74
1,1-Dichloroethene	71
Vinylchloride	65
Carbon tetrachloride	63
Chlorobenzene	62
1,2-Dichloroethane	61
Nickel and compounds, NOS (Ni)	59
Heavy metals, NOS	57
Pentachlorophenol (PCP)	52
Naphthalene	45
Trichloroethane, NOS	40
Methyl ethyl ketone	39
Volatile organics, NOS	34
Manganese and compounds, NOS (Mn)	29
Iron and compounds, NOS (Fe)	28
Barium	28
Acetone	27
Chromium, hexavalent	26
Arsenic and compounds, NOS (As)	25
Dichloroethylene, NOS	25
Phenanthrene	25
Benzo A pyrene	25
1,1,2-Trichloroethane	24
Anthracene	21
Styrene	21
Pyrene	21
Creosote	21
DDT	21
Sulfuric acid	20
Selenium	20
Lindane	20
Tetrachloroethane, NOS	19
1,1,2,2-Tetrachloroethane	19
Waste oils/sludges	19
Bis (2-ethylhexyl)phthalate	18
Acid, NOS	18

TABLE 4. *(Continued)*

Substance Name	Site Frequency
Fluorene, NOS	17
Benzo (J,K)fluorene	17
Radium and compounds, NOS (Ra)	16
Ethyl chloride	16
Aluminum and compounds, NOS (Al)	14
Dichloroethane, NOS	14
Trichlorofluoromethane	14
Dichlorobenzene, NOS	13
Uranium and compounds, NOS (U)	13
Acenapthene	13
Trinitrotoluene (TNT)	13
Chlordane	13
Radon and compounds, NOS (Rn)	12
Cis-1,2,dichloroethene	12
Cis-1,2-dichloroethylene	12
Cis-dichloroethylene	12
1,2-dichloroethylene	12
Asbestos	12
Tribromomethane	12
Antimony and compounds, NOS (Sb)	11
Hydrocarbons, NOS	11
Chloromethane	11
Di-n-butyl-phthalate	11
Hexachlorobenzene	11
Tetrahydrofuran(I)	11
DDE	10
Dioxin	10
Ammonia	10
Dieldrin	10
Chrysene	10
Cresols	9
2,4-Dinitrotoluene	9
Hexachlorocyclopentadiene (C56)	9
Methyl isobutyl ketone	9
Waste solvents	9

GLOSSARY OF ACRONYMS

ARAR - Applicable or Relevant and Appropriate Requirements

CERCLA - Comprehensive Environmental Response, Compensation and Liability Act

EDD - Enforcement Decision Document

FS - Feasibility Studies

HRS - Hazard Ranking System

MCL - Maximum Concentration Limit

MCLG - Maximum Concentration Limit Goal

NCP - National Contingency Plan

NPL - National Priority List

PRP - Potentially Responsible Parties

QC - Quality Control

RA - Remedial Action

RCRA - Resource, Conservation and Recovery Act

RD - Remedial Design

RI - Remedial Investigation

RI/FS - Remedial Investigation/Feasibility Study

ROD - Record of Decision

SARA - Superfund Amendments and Reauthorization Act

SI - Site Inspection

SITE - Superfund Innovative Technology Evaluation

SWDA - Solid Waste Disposal Act

TSCA - Toxic Substances Control Act

USEPA - United States Environmental Protection Agency

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ENVIRONMENTAL IMPACT ASSESSMENT OF INDUSTRIAL WASTES ON SURFASE AND UNDERGROUND WATER IN URBAN CENTRES OF WEST AFRICA:
THE EXAMPLE OF NIGERIA.

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Summary;

An attempt is made to examine the characteristics of the pollution with particular reference to the distributional characteristics of the wastes produced by industries. The study illustrates these pollution characteristics using industries in some urban centres of Nigeria, examines and discusses the environmental effects of the wastes particularly on surface and undrground water, and discusses the waste and water management acpects of the urban centres.

The results of the study show the need to emphasize data collection on industrial wastes. It also emphasizes the need for studies on their environmental impacts particularly on water resources. The study further indicates the need for research and commission studies on the problems of wastes and the possibility and/or feasibility of solution to the problems they create. The study finally emphasize the need. (a) for intensive education and public enlightment on wastes and their impacts; (b)

to get solution to the problems created by solid wastes; (c) for co-ordination and co-operation of the ectivities of and at all levels of government and (d) to examine the possibility and/or feasibility of the transfer of technology particularly for the purpose of solving the problems eccociated with pollution and for the possibility of making projection into the future needs of the country in general and the urban centres in particular.

INTRODUKTION:

The post-independent period in Nigeria has witnessed rapid growth and development through industrialization. Paticularly in the 1970s, the benefits accruing to Nigeria from her mineral resources exploration and exploitation, especially petroleum resources, led to rapid indrease in volume of industrial, commercial and constructional activities. These, in turn have led to rapid development of many Nigerian cities, the creation of new ones and the rapid increase in the size of the population of the urban centres; which in turn have led to phenomenal increase in the volume and diversity of wastes generated daily. All these have created very serious environmental problems with their consequent effects on water, air and land, in addition to the hazards to health and other social and economic ectivities. In many cases, such hazardous wastes contribute to serious illness, and in some cases, death.

For example, industrial wastes in uran centres of Nigeria have profound impacts on surface and underground water, for example through increasing deterioration in quality of these sources of water. Yet, both surface and underground water are important sources of water supplies in many of the cities. This is for example the case in Lagos where many industries and households are becoming more and more dependent on groundwater as a major source of water supplies for their needs. For example, in a recent study by the Federal Ministry of Housing and Environment in Nigeria, on water use by industrial establishments in Lagos Metropolitan area, 31 of the 39 industries selected for the survey report (i.e. about 80%) completely depended upon boreholes or wells for their water supplies while four other industries (another 10%) depended upon boreholes bur also supplement their daily water needs from public water works. One of the industries (i.e. 3%) received its water supplies from tankers (usually drawn from the main public water works) while only 3 industries (about 7%) replied mainly on public water works. Moreover groundwater has become a major source of domestic water supply in many urban centres as well as most rural areas of the country.

The magnitude of the waste problems has created considerable awareness particularly from the governments of Nigeria. In spite of the increasing awareness, not much has been achieved in assessing the characteristics of pollution in the urban centres of Nigeria, and in getting solutions to the problems created by the various consequences of the pollution.

In the present paper, therefore, an attempt is made to examine the characteristics of the pollution with particular references to the distributional characteristics of the solid wastes. The study particularly emphasizes solid wastes characteristics produced by industries, which are known to be, by far the largest source from where these wastes are generated. The study also illustrates the pollution characteristics using some industries in some urban centres of Nigeria, examines and discusses the environmental effects of the wastes particularly on surface and underground water and discusses the waste and water management espects of the urban centres.

THE STUDY APPROACH AND STUDY LOCATIONS.

In this study, both direct and indirect sources of industrial pollution will be examined. The direct sources refer to sources associated with the factories or industries themselves. The indirect sources refer to sources outside the premises of the factories or industries. For example, unlike in edvanced countries, urban centres in the most developing countries in general and Wast Africa in particular are well noted for indiscriminate dumping of both domestic and industral refuse. In many cases, such refuse dumps have approached the size of large mounds or hills and usually, they remain uncollected for several months on the same site. Even in cases where the state wate management organizes dump sites, these are hardly prepared os proper sanitary land fill sites as they should, in order to avoid underground pollution since most parts of the urban centres have rather shallow underground water table or acquifers. The dumps are significant sources of a lot of leachate, a very smelly, coloured liquid, produced decomposing refuse comes in contact with water such as underground water or during the rainly season. Some of the wastes producing the leachate particularlu those from industrial sources are toxic and hazardous to human health and natural resources. Other examples of indirect sources of industrial pollution in urban centres of West Africa include agricultural runoffs which are sometimes significant for containing pesticides.

As in many other parts of the world, the urban environment of West Africa consist of mosaic characteristics reflecting different landuse characteristics, ecological conditions, levels and variety of economic activities and consumption styles and systems of cultural values. The complexity of the characteristics make it difficult to have representative urban centres which will be completely satisfactory for any study on the environmental impact assessment of industrial wastes on surface and underground water. However, based on the degree of industrialisation, four basic categories of cities can generally be recognized in West Africa. These include, very highly industrialized, highly industrialized, moderately industrialized and poorly industrialized urban centres.

For illustrations in this study, examples of the first two categories of cities have been used. These cities include Lagos, Kaduna and Port Harcourt for very highly industrialized cities and Kano, Ibadan and Jebba for highly industrialized urban centres. Of course, the level of industrialization refelcts the level of urban development and to a large extent, the size of the total population and its distributional characteristics in the urban centre. For example, the six urban centres used in the present study are millionaire cities. They reflect considerable spatial developments and rapid changes in land use characteristics over the past two and half decades.

CHARACTERISTICS OF INDUSTRIAL WASTES

Industrial solid wastes considerably contribute to environmental pollution effects in many of the urban centres of West Africa. As already noted above many of these wastes are usually found as indiscriminate refuse dumps, which have approached the size of large mounds or hills and which usually remain uncollected for several months on the same site. The characteristics of these wastes vary in time and space over the urban centres as well as from one urban centre to another.

For example, the mean percentage composition of some components of solid wastes in the urban centres used in this study shows that industrial remnants from paper, tin and metals as well as from bottles and glasses form relatively high percentages of solid wastes around the urban centres. These three components of the industrial wastes are in addition to the other industrial waste components for example part of the food remnants. The figure shows a lot of variations from one urban centre to another. The mean percentages of the food remnants are relatively higher in Port Harcourt than in Lagos while the percentages of paper, tin and metals as well as bottles and glasses are higher in Lagos than in Port Harcourt. a more detailed spatial analysis of the intra-urban variations in Lagos shows that the largest proportion of food remnants occurred in Ikeja area while the largest proportion of tin and metals as well as bottles and glasses occurred in Surulere area of the urban centre. The largest proportion of paper occurred in Mushin part of Lagos. Similar comments can be made for the other urban centres. In all these cases the proportions for food remnants are highest.

A lot of changes can also occur in the components of solid wastes in an urban centre. For example, food remnants formed about 9% of the solid wastes composition in the old town and about 70.3% in the new town of Ibadan in 1969. By 1982, these proporsions have decreased to 6.7% and 27.4% respectively. Similarly, bottles and glasses formed only 0.5% and 2.5% respectively ion 1969 whereas they formed about 6.1% and 10.9% respectively in 1982. It may be finally noted that in addition to the solid wasted indiscriminately dumped in many areas of the urban centres, a lot of refuse dumps are also sometimes found around the industrial centres. All these, in addition to the liquid industrial wastes from the factories significantly affect both surface and underground water in the urban centres.

The characteristics of the liquid effluents however vary from factory to factory. For example, in a study of the effluents from the Paper Mill in Jebba, the composition of total solids was 675 mg/litre, while the values for the dissolved and suspended solids were 204 mg/litre and 471 mg/litre respectively. Calcium formed about 1.6 mg/litre and magnesium about 0.1 mg/litre. In a sample of combined factory effluent of the Nigerian Breweries Limited (NBL), it was discovered that the composition of the total solids was 2,530 mg/litre while the value of some other components are sodium (40 mg/litre), calcium (9.0 mg/litre), potassium (28 mg/litre), zinc (1.7 mg/litre), copper (0.2 mg/litre), and manganese (0.5 mg/litre). The pH was only 4.6. In the case of the Guiness Nigeria Limited Lagos, the values of the concentration of the pollutants included suspended solids (914 mg/litre), while the pH was 8.6. In one of the textile factories examined in a urban centre in Nigeria, the value of the total solids was

3,510 mg/litre while calcium, manganese and magnesium were 0.2, 1,000 and 0.3 mg/litre respectively. All these lquid effluents can no doubt significantly affect both surface and underground water and the health of the inhabitants of the cities who depend on the water. In the following discussions, examples of these effects will be discussed.

ENVIRONMENTAL IMPACT OF INDUSTRIAL WASTES

(a) Surface water.

An example of environmental impact of industrial pollution on surface water can be illustrated with the study of Nigeria Paper Mill (NPM) in Jebba referred to above. In his study, it was found out that a major offending pollutant is coloured settleable solids (pulb fibre), and these precipitate into River Niger a few metres downstream of the discharge point. However, because the volume of the effluent is small compared with the volume of the river, the adverse effects of the effluents are localized to the immediate vicinity of the discharge point.

The pH from combined factory effluent was relatively low (4.4); consequently the pH downstream of river Niger was reduced. Similarly because the total solids from the combined factory effluents were high, these pollutants were increased downstream of the river, particularly within a distance of about 50 metres from the discharge point. Other industrial waste components that were also increased within this range because of the effects of pollution from the mill include dissolved solids,suspended solids, magnesium, sodium, manganese and iron. In general, however, the effects of the effluent discharge on the water quality appears minimal due to the high dillution rate of the river water. Because of this, one may be tempted to ignore whatever waste management problems may exist in the factory. However, because of the rapid developments which are taking place in the factory it is important to ensure a very good management of the industrial wastes and a surveillance at the factory to monitor the quality of the effluent discharge into the river.

The environmental impact of industrial pollution on surface water can also be illustrated with the effects of textile industries in Kaduna, Kano and Lagos. In all these cases, the effects on the receiving rivers and streams include colouration of the streams and river water, hogh pH, high total solid contents and relatively high values of sodium, magnesium, calcium, manganese and iron. If one realises the fact that it is the common practice among a high proportion of the population in the country to depend upon streams and rivers for their daily water requirements, one will definitely realise the need to ensure to monitoring of the effluents discharged into these streams and rivers. For example as noted by the FMHE (1982), the inhabitantsof Nassarawa, near Kakuri industrial estate in Kaduna depend mainly on the polluted river or shallow wells dug along the banks of the river for their needs.

On May 6, 1979, the Sunday Times of Nigeria carried a story which was captioned "Danger Feared Over Water From Estate". This story was referring to the Ikeja Industrial Estate of Lagos. According to this article, the cause of the danger was the hot stinking water which gushed out in different colours from the Wemabod Effluent Treatment Plant already referred to above.

The effects of the effluents of this plant were devastating on both surface and groundwater. For example, the waste water, discharged into the Shasha and Iyaalaro streams of the Industrial Estate, was heavily polluted and this had a lot of environmental impact on the characteristics of some quality parametres of the water in both streams. Indeed, as perceived by the inhabitants living around the streams, the colour of Iyaalaro stream normally changed daily, hence, the name "Iyaalaro" (the mother who dyes) sream. This sream as well as many others in the industrial estates of Lagos Metropolitan area, join the network of creeks and empty into the Lagos Lagoon. Thus, the environmental effects of the pollutants on surface water are usually felt far away from the point of discharge of these pollutants.

(b) Underground Water.

As for surface water, industrial pollution has a lot of environmental consequences on underground water. For example in a study of the effects of the effluents from Wemabod Treatment Plant referred to above, many wells were so polluted that they could no longer produce potable water to the residents of the Ikeja Industrial Estate. Indeed som of the wells were blocked by earth moved into them by the force of the waste water from the effluent.

Recently, a study was carried out on the quality or possible contamination of underground water around areas close to refuse dumps in Lagos Metropolitan area. The areas for investigation were carefully chosen so that there were existing wells (boreholes) for underground water supplies to the residential or industrial establishments around them. Fig. 1 shows some resultys of the study. The upper limit of concentration recommended by the World Health Organization (WHO) in the International Standards for Drinking Water for Mercury is 0.001 mg/litre; iron 0.01 mg/litre; and copper 1.5 mg/lire, based on an assumed daily intake of 2.5 litres per person weighing 70 kg.

As can be noted in figure 1 for the four metals, the water wells (boreholes) around the refuse dumps have relatively high concentration of the heavy metals, and in many cases, higher than the WHO International Standards.

For example, the water wells around Lawanson have higher concentrations of iron and mercury in all the locations whose groundwater was examined. The water wells in some of the locations also have higher concentrations of manganese. Similar results were obtained for the industrial area at Ikeja with the water wells having higher concentrations of iron and mercury that the WHO Internatinal Standards. Comparatively, and surprisingly, however, the Ikeja Industrial area has relatively lower concentrations of these metals than the Lawanson (residential) area of the metropolis. Assuming that in general, Lagos Metropolitan areas have relatively similer natural concentrations of these metals, the study clearly shows the relatively significance at the effects of refuse dumps on surface and underground water in the urban centres of West Africa.

The results of the study no doubt shows examples of the potential risks to human health posed by groundwater contamination particularly by heavy metals and toxic chemicals. There is no doubt an urgent need to improve the standards of the groundwater supplies.

CONCLUSIONS

The significance and the need for impact assessment of industrial wastes on surface and underground water arises from the fact that water is one of the three most important needs of life., the other two being air and food. In most urban centres of West Africe, groundwater and surface water are still the most widely used and the cheapest sources of water. Unfortunately industrial pollution can, and has indeed rendered whatever groundwater and surface water is available to the community unsuitable for drinking or other uses in many areas. To drink them can produce dangerous gealth effects. Even when treated, polluted water can impart unacceptable taste and odour.

Thus, as industrial development occurs and technology is exploited more and more to meet the day rewuirements or increasing sophisticated societies, water resources must be given adequate protection against pollution, otherwise man will continue to be faced with the problems of water resources in West Africa.

The problems of environmental impact of industrial wastes on surface and underground water in the urban centres of West Africa in particular and the urban centres in the tropics in general also raises a number of issues, for example with regards to the management and disposal of wastes, the organizational characteristics of wastes and the problems of technology on the processing and disposal of wastes. For example, studies in some Nigerian urban centres have shown that industries in these urban centres normally choose the cheapest (not necessary adequate) method of waste disposal. It is very common to see that the premises and the immediate environment of many factories are very filthy, being littered with solid wastes of all kinds especially the non-process wastes. Many industrial establishments contract out the disposal of their solid wastes to private contractors, who dump these wastes indiscriminately, at any convenient sites, usually far from ideal. Where refuse bins are provided around the factories or in different parts of the cities they are usually overflowed with reuse indicating the results of irregular collection and bad practice by people who dump refuse on the ground rather than inside the bins.

In most industrial centres waste water are discharged from industries untreated, on land or water bodies, directly or through open sewer with complete disregard for the environmental consequences of this practice.

To save the situation, it is important that the developing countries in general and the West African countries in particular must develop and implement environmental pollution control strategy that is best suited to the environment. Such a strategy must involve intensive education and public enlightenment on wastes and thei impacts, co-ordination and co-operation of the activities of government at all levels, and examination of the possibility of the transfer of technology particularly for the purpose of solving the problems associated with pollution and for the possibility of making projection into the needs of the various countries of the region in general and the urban centres of these countries in particular.

The paper no doubt shows the need to emphasize data collection on industrial wastes in any adopted strategy. It also emphasizes the need for studies on the environmental impacts of the wastes, particularly on water resources. There is no doubt the need for constant monitoring of the effectiveness of any pollution strategy that may be adopted if developing countries in general and the West African countries in particular are to bequeathe "clean water" to the future generations.

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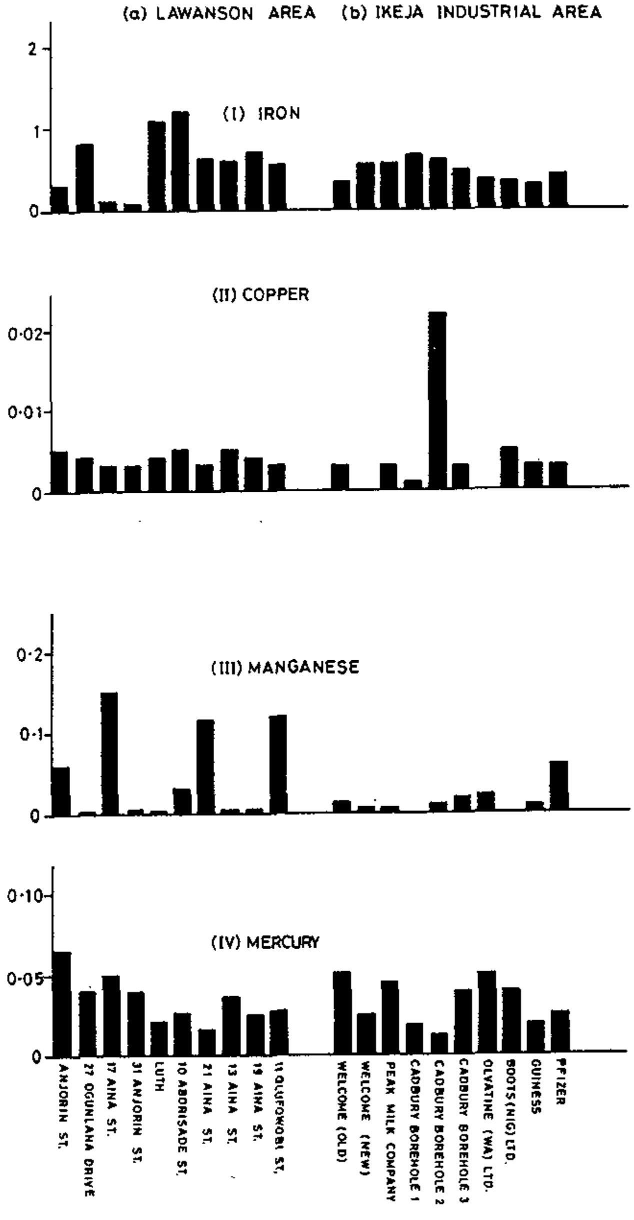
(a) LAWANSON AREA
(b) IKEJA INDUSTRIAL AREA
(I) IRON
2
1
0
(II) COPPER
0·02
0·01
0
(III) MANGANESE
0·2
0·1
0
(IV) MERCURY
0·10
0·05
0
ANJORIN ST.
27 OGUNLANA DRIVE
17 AINA ST.
31 ANJORIN ST.
LUTH
10 ABORISADE ST.
21 AINA ST.
13 AINA ST.
19 AINA ST.
11 OLUFOWOBI ST.
WELCOME (OLD)
WELCOME (NEW)
PEAK MILK COMPANY
CADBURY BOREHOLE 1
CADBURY BOREHOLE 2
CADBURY BOREHOLE 3
OLYATINE (WA) LTD.
BOOTS(NIG) LTD.
GUINESS
PFIZER

WHY ATTACK UNDERGROUND CHEMICAL POLLUTION BY INTENSIVE AND HIGH-TECH REMEDIAL MEASURES, WHEN NOT NECESSARY?

3 CASE-STUDIES.

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Summary.

A vast amount of underground chemical pollution cases have turned up in the industrialised countries during the last decade. This calls for action, and lots af techniques are developing for treatment of this pollution. However, the costs of a high level clean-up are tremenduous in many cases, and thus investments in thorough investigations of environmental impact risks may prove advantageous in individual cases. This is even more true as many remedial measures do not actually remove the pollution; it is merely moved from one environmental sector to another. 3 cases, in which the remedial measures have been designed on a rather low-level, according to a judgement of environmental risks, are presented.

INTRODUCTION.

Ground water and surface water pollution from chemical dump sites have become a major environmental action field in the industrialised contries during the last 5-10 years. Lots of money are invested in technological development and spent on investigations and remedial measures, and for good reasons; the knowledge of many aspects of the problems is rather limited and a thorough clean-up of a chemical dumpsite including polluted soil and water is often enormously expensive today.

Further, a high level remedial measure demands large amounts of energy, and is thus an environmental problem in itself. In some cases it might even be argued whether the total environmental effect of such a procedure is positive or negative!

Therefore, from both economical and environmental viewpoints, there are good reasons to consider carefully the induvidual case, in order to design the action-level according to the actual environmental risks and not to some, more or less arbitrarily chosen, "environmental standard". But can this be done on safe ground today? The answer to this queston is undoubtedly "no", and this is one of the reasons why a lot of different ways of approach have developed in different countries. There are of course other reasons: Heavily industrialised countries tend to deal with the problems "the industrial way", i.e. by creating firm administrative procedures and building "cleaning industries" etc.; tradition plays a role too.

In Denmark there are several trends, but generally it is considered to be a sound principle to design the clean-up measures according to a careful judgement of the potential environmental conflicts. One of the reasons for this is limitation of costs; a good reason, considering the present situation of national economy in Denmark. A maybe better reason is a wish to deal with the problems in a reasonably rational manner. Of course, it is recognized, that lack of knowledge about many specific aspects might produce some insufficient solutions. However, we believe that the "environmental conflict design approach" is the best one in the long run, and that the only way to learn it, is to use it.

In our company we have been using this concept from the start, and intend to keep on developing it in the future.

In the following, 3 very different examples of applying the concept are presented; the reader way then judge for himself, whether we are good at it or not!

CASE I: Coal tar pollution at Uldum, Vejle Province.

Pollution source

The pollution source is a factory for road materials production, using coal tar as a raw material from the 1890's to about 1970. The main site is the factory area, and the coal tar pollution is mainly concentrated around 3 abandoned and sand filled tar pits in the NW part of the area. The pollution in the ground has been caused by slow leakage from the pits, and the total amount of coal tar present as underground pollution has been judged to about 50 tons. Localisation map fig. 1.

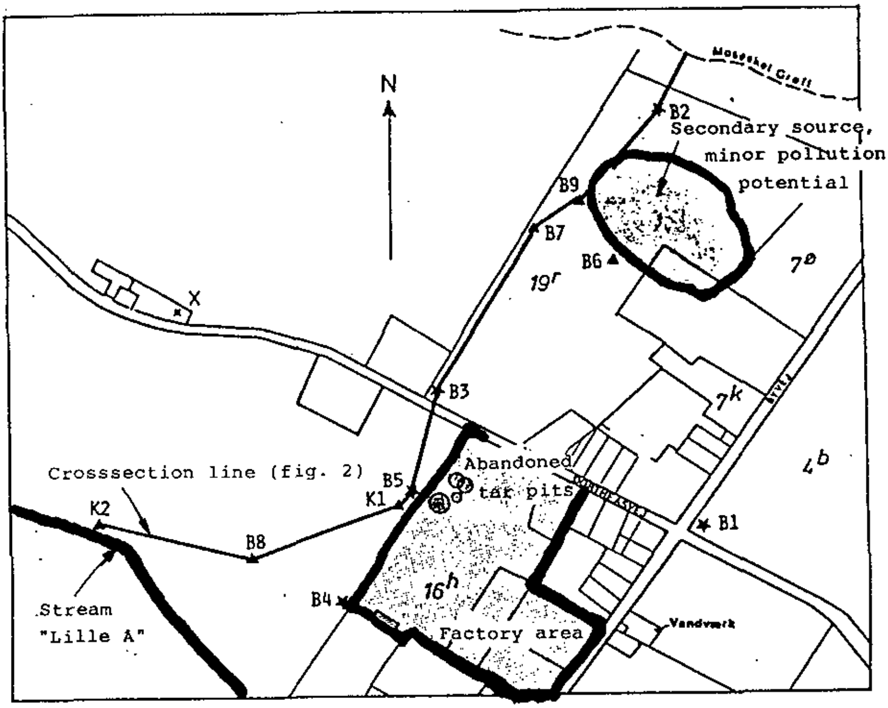

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Fig. 1 Map of source area. K's and B's are investigation boreholes.

Hydrogeology

The hydrogeologic conditions are shown on the cross-section in fig. 2 (ref. cross-section line on fig. 1). Concerning the main source around the pits, the left hand part of the cross-section (K2-B8-K1) is referred to. There are two ground water reservors in the area, both consisting of rather fine sand; the upper of partly glacial partly miocene origin, the lower of miocene age. The seperating aquitard is rather thin, but contiuous, and of low permeability (documented through borehole logs, *pumping* tests etc.). No significant water circulation is anticipated below the lower aquifer.

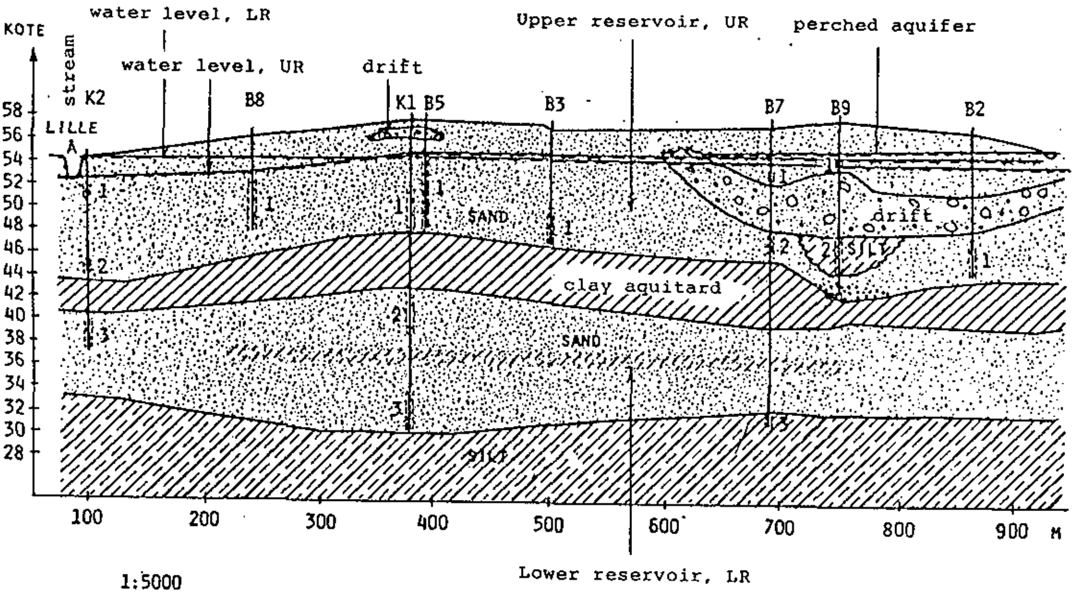

Fig. 2 Cross-section through wells K2-B8-K1-B5-B3-B7-B9-B2.

Ground water flow

The ground water potential of the upper aquifer is shown on fig. 3. It follows, that the flow direction from the pit site (at B5/K1) is towards the west, joining the stream "Lille Å" at a distance of about 300 m's. The flow direction in the lower aquifer is N-NW towards the main stream "Gudenåen" to which "Lille Å" is a tributary. The leakage conditions between the two reservoirs are as follows: There is a downward but faint potential gradient at the pit site. Very shortly west of the site, however, the conditions change to upward gradient (ref. fig. 2). These conditions have been observed during a considerable period and are judged to be persistent. Thus percolation of pollution to the lower aquifer is only possible in the ultimate vicinity of the source.

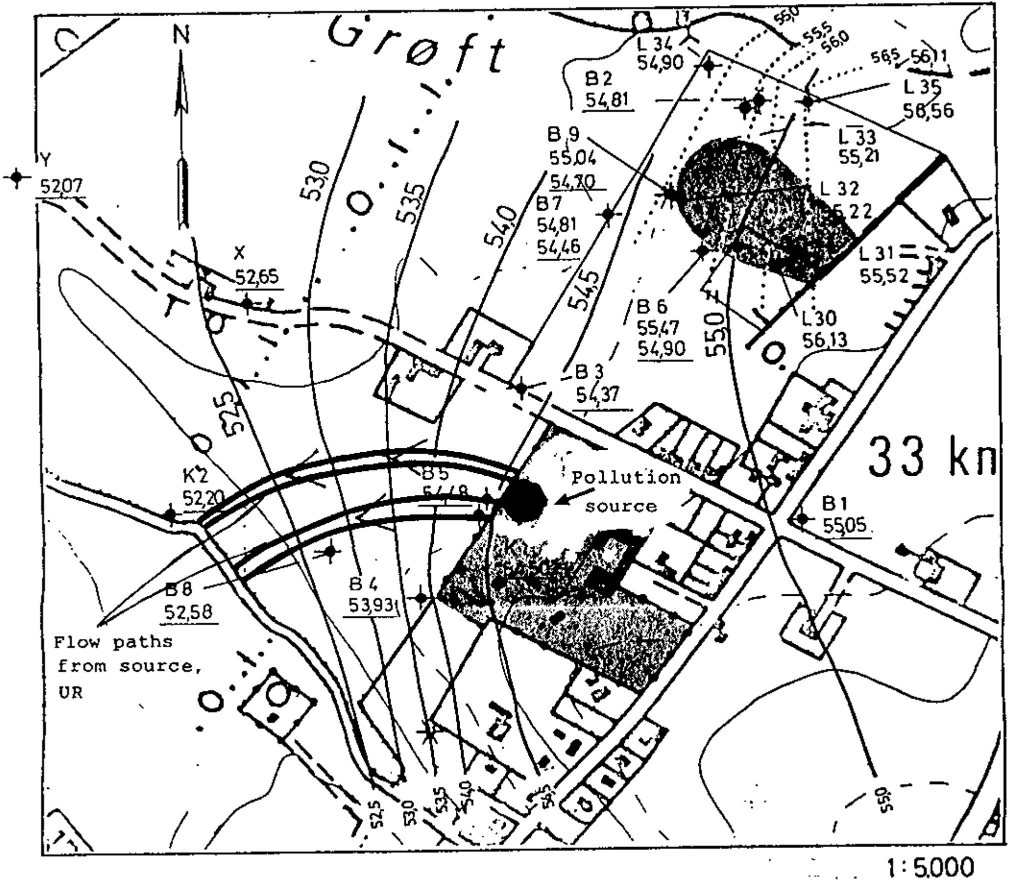

Fig. 3 Map of ground water potential in upper aquifer.

Levels in m's above sea level.

Pollution extension. The spread of the pollution in the ground water is mapped by analysis of phenols, which are the most conservative of the outwashed components from the tar. Also naphtalene, toluene and ammonia has been measured and found in the area. The concentration of phenols in water samples in the upper reservoir are briefly stated: Sample from pit area: 190 ppm; samples from B5: 39-52 ppm-(center of plume); samples from K1: 1,6-7,3 ppm (rim of plume); B8: 0,82 ppm; K2: below 5 ppb. These results confirm the spread of pollution as indicated on fig. 3. The ground water velocity shortly downstream of the source have been estimated to 50 m pr. year and the average phenol concentration to 50 ppm. Combined with the various geometric factors, this results in a judgement of the phenol transport of about 50 kg/year shortly downstream of the source.

Effect in stream. After full break through to the stream, this transport will theoretically produce concentrations here with a maximum of 10 ppb (dilution considerations). Several measurements in the stream have produced results of 0,13-0,78 ppb, which is much less and considered as background level. The conclusion is, that full break through to the stream has not occured and may be never will.

Effect in lower reservoir Water samples from the lower reservoir have been analysed for phenols as well. In K1 indications have been found, but highly dependent on pumping time. Later, additional wells have been drilled, and a comprehensive sampling programme has indicated a phenol-pollution in the lower aquifer, but of very minor intensity (less than 1 ppb, when pumping 10 m^3/hour).

Conclusion Under the present circumstances, the pollution is only present inside a limited area in the upper reservoir, in which no water ressources utilisation takes place. No effect in the stream has been seen, and no serious effect will probably occur. A minor effect in the lower reservoir is present, but not considered serious, and will not develop as long as the leakage conditions are unchanged. No major water ressources utilisation in the lower aquifer is taking place or planned in the future. A cleaning of the site will be very costly (25-50 mill. d kr.) and does not seem justified by the potential environmental effect from the site. Therefore no remedial actions are suggested in spite of the large amount of coal tar present, and the high concentrations of phenols locally in the upper reservoir.

Monitoring

A monitoring programme has been designed and is now running. The pollution development in the upper reservoir in a certain distance from the source is monitored in B8, and the future effect on the stream judged accordingly. The effect in the lower reservoir is measured through sampling from K1 and an additional well K3, downstream of K1 in the lower reservoir. So far no serious development has taken place. The leakage condition is monitored by registration of water levels in the wells, and finally water samples from Lille Å are analysed.

The monitoring activities are scheduled once a year, some once pr. 3 years only, and are not very costly. Remedial actions will only be carried out if an adverse development of the present conditions occur, or if new, and much more cost-effective cleaning methods are developed.

Case II: Pollution from township area, Maribo municipality.

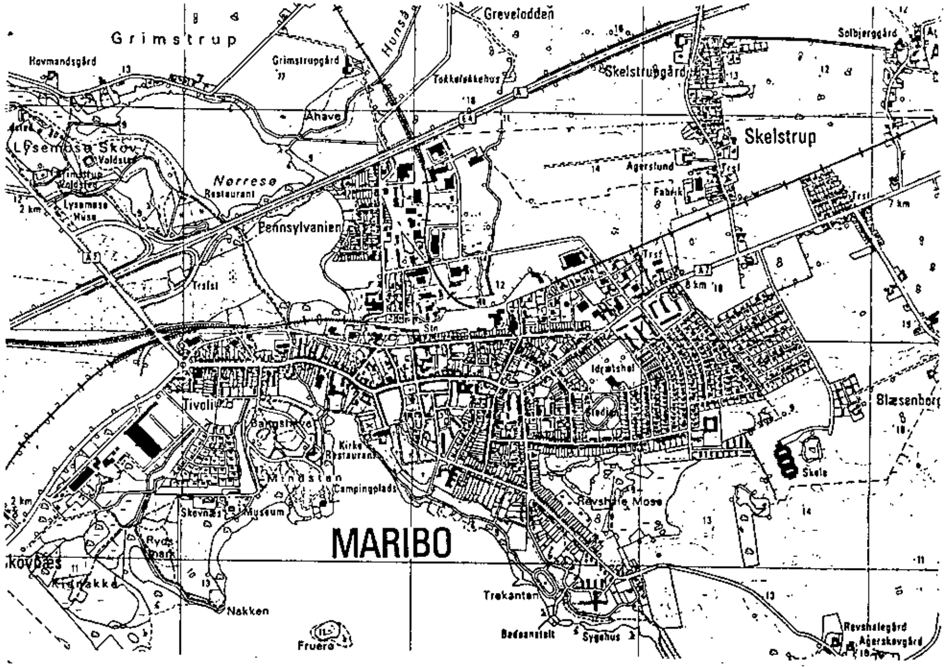

Fig. 4 Maribo township. 1:25.000

Unknown industrial sites.

While starting the detailed investigations of the first site for chemical waste in Maribo (the gasworks), it gradually was recognized in Denmark, that sites with existing or former industrial activities of certain kinds were likely to be more or less heavily polluted. Thus, in a township like Maribo (only about 5.000 inhabitants), apart from the few registered sites, an unknown number of additional polluted areas had to be expected.

It seemed obvious, that a high intensity investigation and clean-up at one site at a time, could very easily be the wrong strategy, as no sound overall judgement of the situation could be made from the start. Therefore a systematic registration was made from existing information in public files. Based on these, it was possible to pick out any site, where "risky business" was, or had been, taking place, i.e. areas of <u>potential pollution</u>. These were registered and mapped.

Too many sites

Ground water conflict?

The result was 83 sites in a small township like Maribo. Even if many sites could be expected to be only slightly polluted, the total task of investigating and cleaning up all these sites, seemed economically out of proportions, based on existing experience with similar, induvidual cases. Furthermore, the ground water intake to Maribo was situated inside the township area, so a serious potential conflict existed.

The water quality of the township water supply was checked, but was ok; the aquifer is covered by 30 m's of clay on average, and no pollution has turned up yet and might not do so for many years. However, the problem is there, and must be expected to develop sooner or later.

Strategy against ground water pollution.

Total pumping scheme

The following strategy was decided against the ground water problem, which was considered as the main water pollution conflict: No clean-up measures, in stead the whole township is considered as one pollution source, and the following actions planned over a certain period: A new well field is established at a satisfactory distance from the town. The old well field is used to maintain a constant ground water with drawal in the township area in order to secure, that any water pollution from the town area is pumped towards this well-field.

The "town-water", which is initially clean, is suggested used to sustain the surface water system in the area, including the lakes, which often suffers from lack of water. The hydrogeologic conditions are judged to be able to secure, that the "town-water" will never be heavily polluted, but might become unfit for drinking eventually.

The proposed pumping scheme should of cource be designed and established carefully; the existing ground water catchment area as shown on fig. 5, should be reduced as much as possible, but still cover the town area.

It is believed, that a constant pumping rate from the town area of 35-40 m^3/hour should be sufficient to control the situation safely.

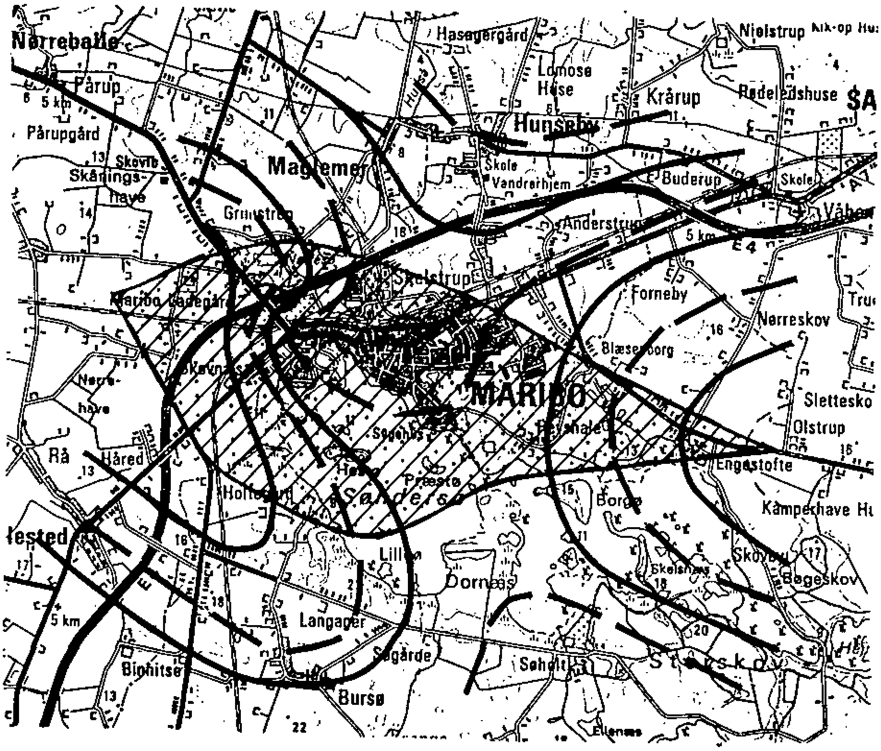

Fig. 5 Present ground water potential and catchment area in Maribo. With a carefully designed pumping scheme from the township, a catchment area about 1/3 of the present should be possible to obtain.

With a use of such a total remedial measure, the ground water problem will be under control. It is hereafter a rather straightforward, though not a simple task, to rank the sources in the town, according to land use risks only, and to deal with them on this basis.

Discussion. Many aspects of the concept may be argued. Basically it is based upon the anticipation, that the hydrogeologic conditions under Maribo will be able to reduce the pollution from the town sources to a level, where the water is acceptable for the surface water system ecology, or maybe even for drinking water.

If this is the case, the concept will save a tremenduous lot of money, and buy us a lot more time! If it is not, the concept might turn out to be equally expensive to the site-after-site approach in the long run. All in all, the "total approach" is considered to be very advantageous in many situations in Denmark, and is expected to be implemented in many townships over the coming years.

CASE III: Hydrocarbon pollution on the "petrol island", Prøvestenen, Copenhagen.

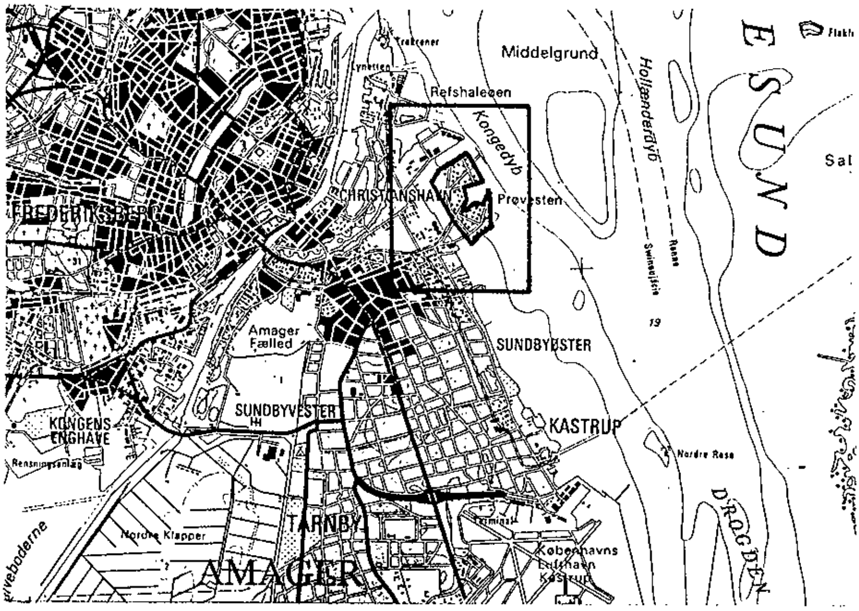

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Fig. 6 "Petrol island", Prøvestenen, location map.

Unexplainable spills.

On the island Prøvestenen in Copenhagen Harbour, the oil companies have for decades kept massive storages of various hydrocarbon products. Small and large spills to the surronding waters have occurred occasionally with some environmental problems to follow. Many spills of course resulted from technical and human errors, but a considerable part was hard to explain.

Surface water drainage.

The island is today covered with oil tanks as shown on fig. 7. The island consists of basins filled with pumped sand, separated by dams, made of more impermeable material. A major part of the island is covered with tarmac and the surface water is drained off by a comprehensive pipe-system. All these pipes pass an oil-separator before they penetrate the perimeter dam. The locations of the oil-separators are also shown on fig. 7.

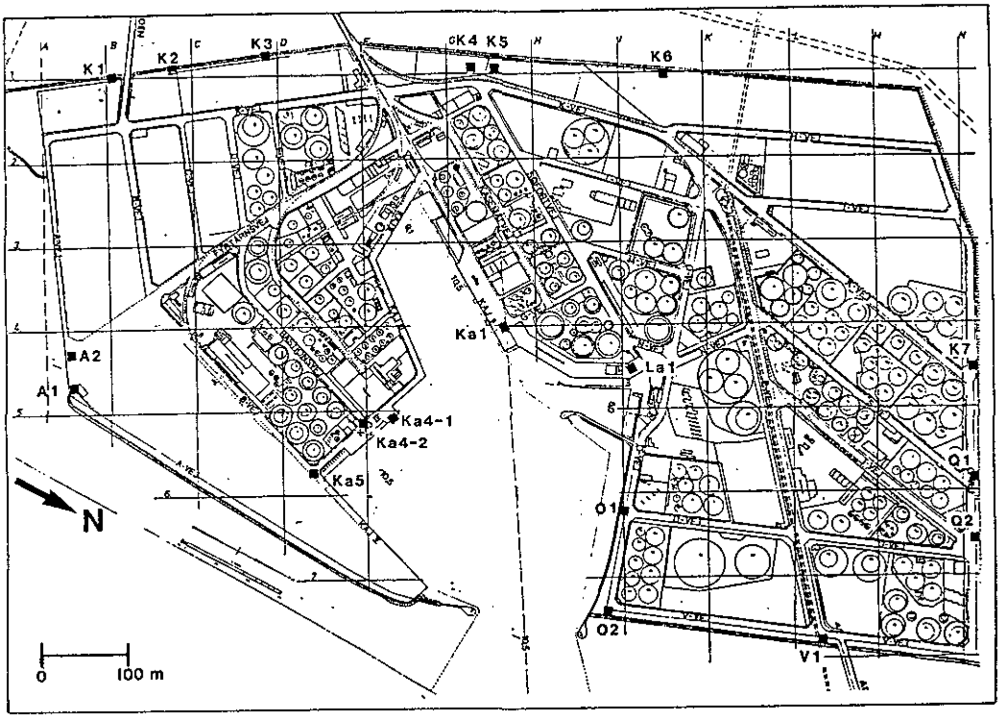

Fig. 7 Prøvestenen, map showing tanks, roads and oil separators on the rainwater system (■).

Oil lenses on the ground water

It was suggested, that the unexplainable spills could be caused by outlets from occurrencies of oil on the ground water. A drilling programme was carried out, and more than 50 piezometers established. In many parts of the island, oil occurencies were encountered. Thicnesses of oil on the ground water up till more than 1,5 m were found.

Rain water pipes as drains.

Then the surface water system was checked. The pipes were found to be old, and probably leaking. Moreover, they were situated below the ground water level on a large part of the island, i.e. functioning as drains if leaks were present. The pipe system would therefore to a considerable extent attract the oil lenses on the ground water, and oil would enter the pipeline under certain conditions.

Oil separation in pipe system

It was further found, that near the coastline, many of the pipes were actually below normal sea level, and for some oil separators both the ingoing and outgoing pipe was more or less permanently under water. Under such circumstances, the oil seperation does not take place in the separator, but in the pipe system upstream, fig. 8. In this situation a sufficient drop in sea water level, will cause the oil in the pipeline to enter the separator in such amounts, that separation can not take place, and the oil will enter the harbour.

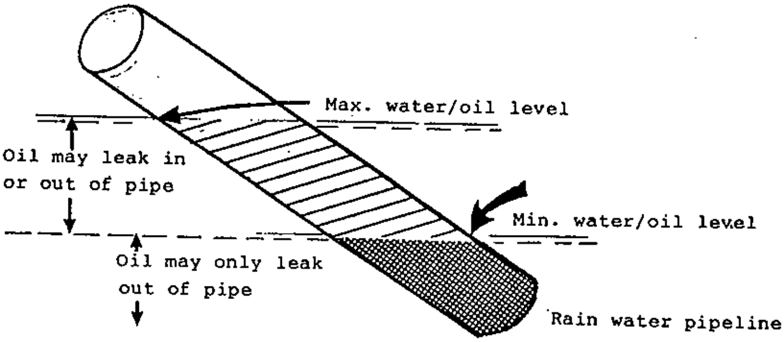

Fig. 8 Sketch of oil exchange between oil lens and pipe, and separation in pipe.

On Prøvestenen, the primary conflict is the spill of oil to the water surrounding the island, whereas the oil lenses on the ground water is a minor problem, as long as the land use is unchanged. The remedial measures have therefore so far merely aimed to reduce the spill from the island. The measures have been many: Better technical procedures and control on the island in general, have had a major effect.

Remedial measure: perimeter control

The spills from the ground water are presently being put under control by establishing hydraulic barriers along the critical parts of the perimeter. These barriers are made in several ways:

- The under water oil separators are put into normal action by installation of a pumping system to keep down the water table;
- drains with pump-fitted collector wells and oil separators are constructed along certain parts of the perimeter;
- other parts of the perimeter, where numerous installations does not allow establishment of drain systems, are equipped with individual pumping wells to establish an efficient hydraulic barrier.

As long as the land use does not change on the island, more intensive measures are not planned for.

CONCLUSIONS.

3 examples have been described, showing different ways of designing remedial measures according to the actual environmental conflicts present. In all cases the results have been measures, which were much less costly than intensive standard-actions, and equally safe for the environment.

We believe, though many uncertainties are present, that the "environmental conflict design approach" will produce much better cost/benefit ratios in the long run, than any other approach. As a side-effect, it will expand overall environmental knowledge much faster, than more rigid decision-making systems.

STRATEGY FOR DISPOSAL OF SPECIAL (HAZARDOUS) WASTE AND DESIGNCRITERIA FOR LONG TERM LANDFILLS OF PRETREATED INORGANIC WASTE IN AUSTRIA

by Karl J. Rohrhofer and Wolfgang Winkler, Vienna, Austria

Summary:
This report deals with the **waste management** in **Austria** with respect to **Lower Austria** and **Vienna** in particular.

Philosophy of safe long term landfills for the disposal of special (hazardous) waste is described. Proposals for **leachate free landfills** are made.

A detailed description of the recently made decision of the **Lower Austrian Model on Waste Management** is given. It shows the structure of the Lower Austrian way of waste reduction / waste collection / waste disposal, the whole matter of waste management.

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1 INTRODUCTION

1.1 **Hazardous waste** as a liquid, slurry or solid from industry and trade that is not properly treated and disposed of <u>can</u> **contaminate the air, water and soil** and <u>can</u> be dangerous for the environment e.g. due to its toxicity, threat of water pollution. Public awareness has grown steadily since late 70s. A considerable effort has been made, particularly by the industrialized nations, to adapt safer methods of waste disposal.

1.2 Concurrently, increased legislative controls have been introduced, stimulating industry and trade, to **reduce** the **generation of waste to a minimum** and to improve waste reutilization.

1.3 There are doubts as to **whether landfilling is a suitable long-term hazardous waste disposal technique**. Unfortunately, there is a program nowadays to rehabilitate the vast number of sites which were developed without prior design or planning, and which are now causing serious pollution problems.

1.4 The **volume of industrial hazardous waste** produced in the member countries of the European Economic Community (EEC) is estimated to total about between 15 and 20 million tons per year (this figure was given in 1981). The annual increase in waste generation in both the U.S.A. and the E.E.C. is between 2 and 4 % (status of 1983).

1.5 While a large number of different processes of waste disposal falls under the category of "treatment" or "pre-treatment" of waste, the **"end of the pipe" disposal technique of waste has to be landfill disposal**. Where else can the waste be disposed of?

1.6 Only a **severe strategy on hazardous waste management**, including the prescriptions for pretreatment of the hazardous waste and the **design criteria for construction and operation of waste landfill disposals** will solve the problematic task with the necessary and required security.

1.7 This report only discusses the landfill disposal techniques of non gaseous and non liquid (pretreated) hazardous waste.

1.8 There are two factors which have to be considered

- the **influence of the chemical properties** of the disposed waste and
- the **way the waste is placed into the landfill** ("wet" or "dry disposal").

Both factors have to be determined and to a certain extent depend on each other.

The **disposed substances may not react on each other**, therefore pretreatment and/or separated disposal are to be requested.

2 SAFE LONG TERM LANDFILLS FOR DISPOSAL OF PRETREATED INORGANIC WASTE

2.1 The Preconditions

The preconditions for Safe Long Term Landfills - e.g. as *low waste technologies* / separated collection / separated transport / master plan for regions / prescriptions by law - are not explained in detail in this report. But these preconditions are listed up later in chapter 3 where the Model of Waste Management of Lower Austria is described.

2.2 Terminology

2.2.1 Before discussing and describing the design criteria for Safe Long Term Landfills (for the deposit of controlled *inorganic special waste only*) the **definitions** are to be determined.

2.2.2 The Austrian Definition of Special Waste

As given in the title of this report, Austria (and also other European countries) have introduced the terminus **SPECIAL WASTE** (in German: SONDERABFÄLL). According to the Austrian standards (ÖNORM) "Special Waste" covers all organic and inorganic waste where the sources are others than households as well as waste which can not be disposed in garbage landfills.

Therefore "Special Waste", includes all waste from industries, trade and workshops and further waste consisting of rubble and similar materials. Contaminated soils are defined as "Special Waste" once they are collected by a licenced company.

Of course, households in Austria are informed not to put toxic materials (e.g. solvents, paints, batteries, waste oil, medicaments) into the garbage. That waste has to be collected separately-*in most cases* the townships take care of their safe disposal. However, waste generated by households, once collected separately is also determined as "Special Waste".

According to the Austrian terminology, "Special Waste" includes both hazardous and non-hazardous materials.

2.2.3 Short / Intermediate / Long Term

Concerning safe landfills it has to be defined whether the landfill (or other kinds of waste deposits) are expected to function as:

- **short term deposit**
- **intermediate deposit**
- **long term deposit.**

The expression "**LONG TERM**" has to influence the designer into considering the **possibility to reuse or recycle** some of the substances of the disposed waste using technologies which will eventually be developed in future. In many cases this will lead to the disposal of different kinds of waste in separate sectors of the long term landfills or in landfills made for a special type of waste only (mono-deposits).

2.3 Criteria for Long Term Special Waste Landfills

2.3.1 Referring to the decisions made in Austria **Long Term Landfills for Special Waste shall not contain materials determined as high toxic and shall not create any harm to the environment, respectively.**

2.3.2 Therefore **the requirements of the future landfill** design **depend on the waste** to be disposed in it.

2.3.3 Of course, the **most important factor** is **the location of the site of the landfill.**

2.3.4 **Extensive investigations** are to be executed before decisions are to be made about a certain location for a Long Term Special Waste Landfill. The results of these investigations in the following fields will strongly influence the assessment:

- topography
- geology

- hydrology[1]
- meteorology
- infrastructure / use of land and water / environment.

These investigations are prior to any further decisions. They will also strongly influence the requirements for the selection of the permitted types of waste which are to be deposited in a safe long term landfill at a certain location.

2.3.5 Nowadays the state of engineering for long term (hazardous) waste landfills is to some extent insufficient concerning the leachate of a landfill.

Most of the guidelines, published by some countries, only require **"to <u>minimize</u> infiltration of precipitation and leachate"**.

2.3.6 However, until now there is no demand for existing guidelines to "**<u>avoid</u> infiltration of precipitation and leachate at all**".

2.3.7 The proposal of the author of this report, K.J.Rohrhofer, describes a system of a **leachate free landfill** for special waste disposal. The system was recently made known to the Austrian Authority of Patents and is expected to become a patent.

2.3.8 **Multi Barrier System as Design Criteria**
Of course, according to the author's proposal the principal criteria for the construction or operation of a long term landfill will be the same as standard engineering practice:

Latest safety philosophy requires **multi barrier systems**: In the case of a single barrier failure the safety of the whole system remains harmless to the environment.

2.3.9 The considerable advantage of the author's idea for a multi barrier system

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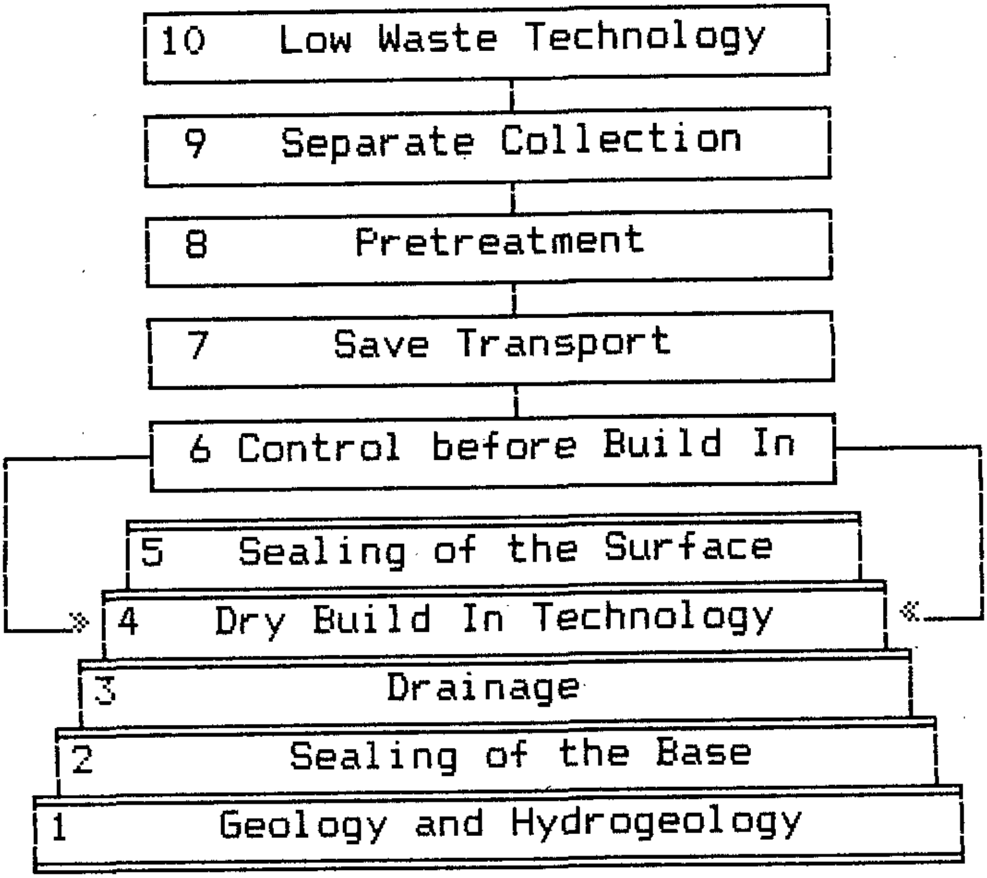
[1] The hydro-geological precondition should be: no (utilizable) groundwater.

has been to add another barrier, namely the **Dry-Build-In Technique** (barrier # 4).

The other barriers, # 1 through to # 10, are containing both the focal points of waste management including waste reduction / disposal and the wall build up of the landfill itself.

2.3.10 The following proposal of a **multi barrier system** for a safe "long-term special waste landfill disposal" gives prominence to the "main" barrier " a **selection of the location** for the waste disposal", # 1.



Structure of a Multi Barrier System for Anorganic Hazardous Waste Disposal

Barrier #1: **Geology** of the location of the landfill disposal, optimal natural tightness and none occurrence of utilizable groundwater are the basic requirements to a long term safety of a hazardous waste landfill disposal.

Barrier #2: **Sealing of the base** of the landfill, to avoid emission of extracts *into ground, if* any at all (see barrier #4 + #5).

Barrier #3: **Drainage**, collection and safe treatment of leachate, if any at all (see barrier #4 + #5).

Barrier #4: **Build-In-Technology**, avoiding extraction caused by water content of the disposed waste (no pressed out water); build-in of pre-treated inorganic materials only, separately disposed for possible later reuse.

Barrier #5: **Sealing of the surface** (and side walls / dams) of the *landfill*, including drainage, to avoid infiltration of precipitation; and to make re-cultivation possible

Barrier #6: **Control before Build-In**, eventual treatment at landfill site to bind remaining hazardous particles.

Barrier #7: **Save Transport** of the (pre-treated) waste to a hazardous waste treatment plant, which may serve as a central place of transshipment for a certain region.

Barrier #8: **Pre-treatment** of the hazardous waste, to grant safe transport and to minimize risk of emission of the finally disposed material.

Barrier #9: **Separate collection** of the waste on-site, where the waste is generated, to avoid later additional treatment.

Barrier #10: **Low waste technology**; the aim is to grant economic emission free waste treatment and disposal.

2.3.11 The materials suitable for **wall build up** of landfill disposals depend on the type or mixture or moisture of the disposed material. Whether the material is suitable also depends on the stress capability and long term behavior of the used material under strain (like strain corrosion).

2.3.12 In any case the build-up of the tightening layers shall be a **combined tightening layer**, a **sandwich compound** according to the state of engineering.

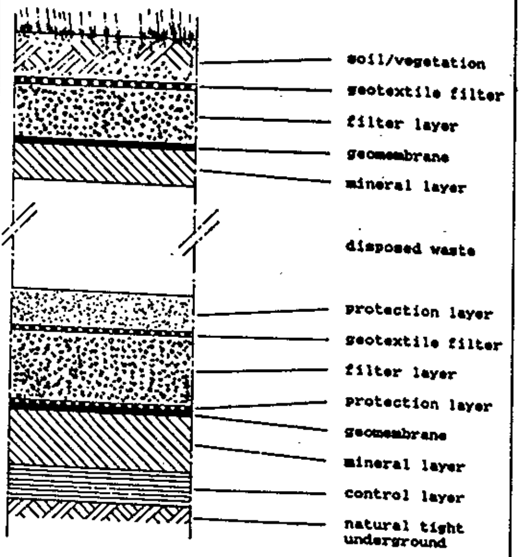
As a rule the **sandwich compound** consists of several **mineral tightening layers and layers of thermoplastic membranes**. Combined with **drains** the **sandwich compound** is to be designed according to the requirements of the upper cover, the side walls / dams and the base of the landfill.

2.3.13 The **drains of the upper cover and of the side walls / dams** of the landfill have to prevent further infiltration of precipitation, if any occurs caused by leaking of the tightening layers.

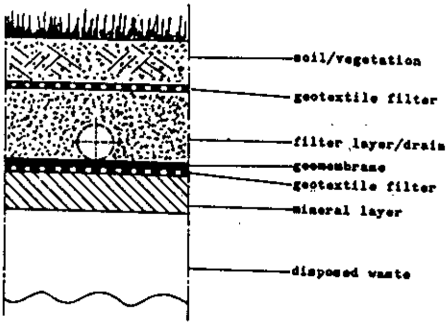
2.3.14 The **drain at the base** of the landfill indicates, whether the side and upper tightening layers, the drains respectively, could keep off any infiltration.

The drain at the base indicates also whether the water content of the built-in waste creates pressed out water or not.

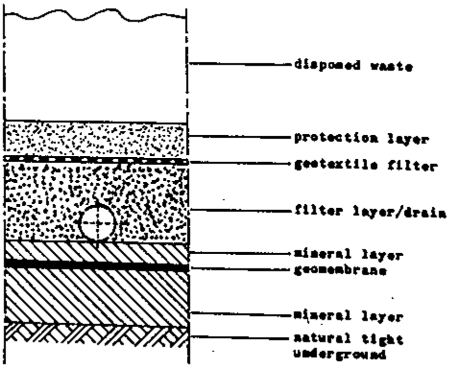
Examples of Sandwich Compound Layers (tigthening and draining)



General Structure of
High Safety Wall Build Up
(Tightening of Surface and Base) of
Hazardous Waste Landfill Disposal



Sealing and Draining of
the Upper Cover and of the Side Walls/Dams
of a Landfill



Sealing and Draining
of the Base of
a Landfill

3 THE LOWER AUSTRIAN MODEL OF WASTE MANAGEMENT

3.1 General Observation

The basic idea of the model of waste management is to **protect the environment**.

Everybody has to learn how to handle the avalanche of waste created by our society. The means are information / education of all citizens to **create deeper understanding** as a consumer, as a trader, as a manufacturer, as a decision maker.

3.2 Elaboration of the Model

3.2.1 Authors of the Model

The model was elaborated by a group of four experts of several divisions of the Lower Austrian Government. It took them approximately two years.

(The authors of the model are in alphabetic order)

- Lambert Führer, grad. Eng. Dr., chemical expert on environmental impacts specifically in the field of air pollution (emissions/immissions)
- Walter Pozarek[*], grad. Eng., expert on regional planning
- Helmuth Siegl, grad. Eng., expert on water and waste management, in particular in the field of waste disposal and landfills
- Wolfgang Winkler, grad. Eng. Dr., expert on environmental impacts specifically in the field of chemistry of water and waste (who is co-author of this report).

3.2.2 History of the Model

The original demand was to find a location for a Long Term Special Waste Disposal. While working at it turned out that a certain location can not be determined without providing a model for waste management.

A group of experts of the Lower Austrian Government compiled a complete model for special waste management, containing low

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This expert was convener of the working group elaborating the Lower Austrian Model.

waste technologies, separate collection, pre-treatment and safe landfill disposal.

3.3 Structure of the Model

3.3.1 The head of the division "Water and Waste Management / Technical Officers/ Experts" of the Government of Lower Austria, worked out a brief structure of the Model of the Lower Austrian Waste Management which is shown below.

3.3.2 The aims in determining a Special Waste Landfill for Lower Austria are to reach an **optimum in environmental protection**, both during the operation and preservation of the landfill disposal site, as well as to **minimize the requirements for long term preservation.**

3.3.3 Therefore the **principal preconditions** are to **reduce the generation of waste**, to **recycle waste** as far as possible and to **realize a multi barrier system.**

3.3.4 The Lower Austrian **requirements** to optimize environmental protection **when selecting a location** for a safe long term special waste landfill are:

- - no (utilizable) groundwater
- - defined drainage condition beside and below the landfill site
- - compatibility with nature
- - sufficient traffic infrastructure
- - location of the landfill away from residential areas.

3.3.5 To fulfill the demands mentioned above the following **barriers** are to be foreseen:

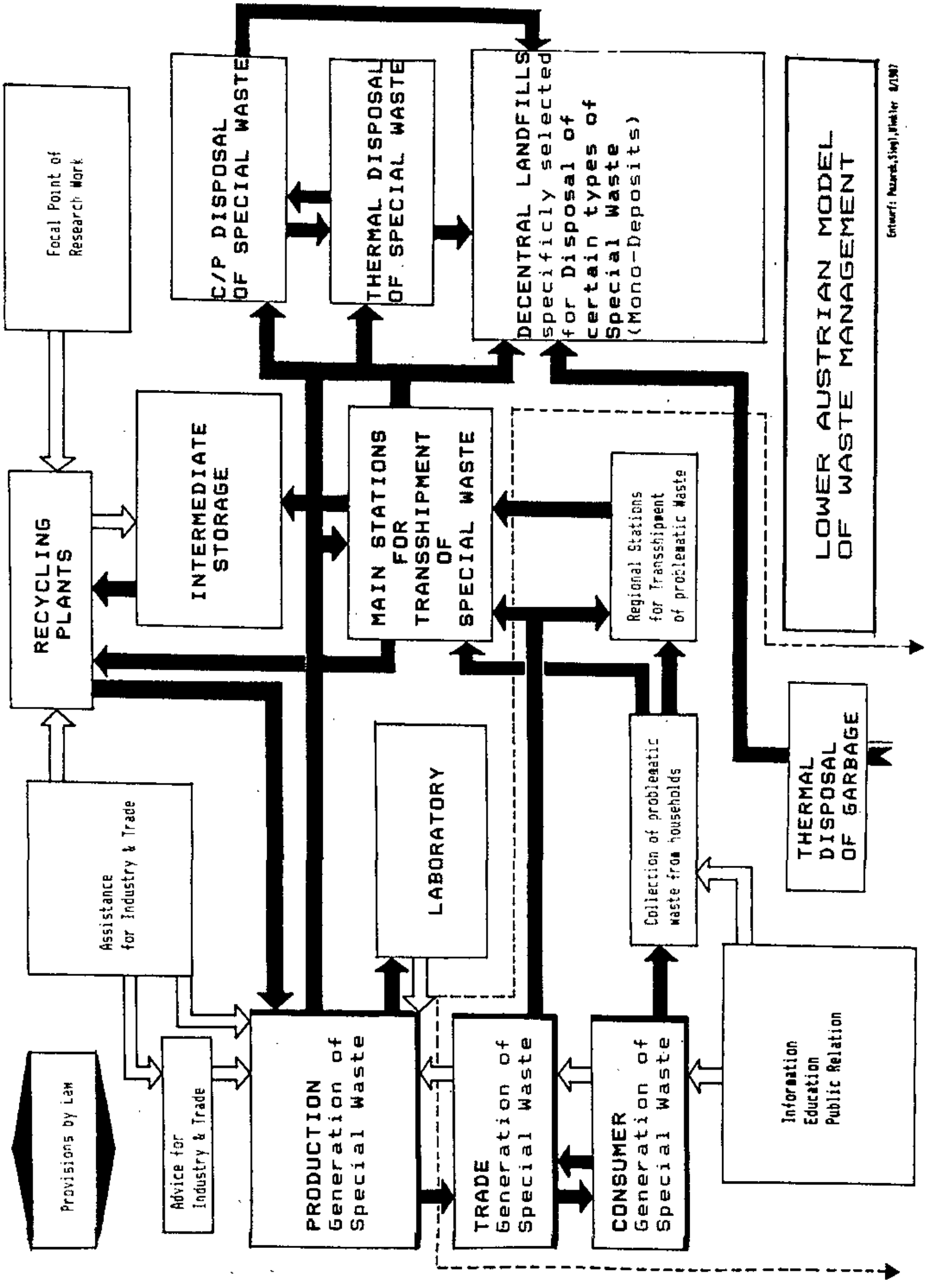
- **technical barriers** of special waste landfill
- - tightening of the base
- - tightening of the upper cover
- - safe disposal of leachate, if any

- **operational barriers** of a special waste landfill are
  - installation as mono deposits only
  - check of incoming waste
  - build-in technology

- **logistic barriers** of a special waste landfill are
  - installation of suitable collecting stations
  - installation of suitable temporary transshipment stations
  - use of suitable means of transportation.

3.3.6 The demands to **minimize the necessities of long term preservation** according to the Lower Austrian Government are:

- disposal of such waste only which can not be further reduced by means of low waste technology and for recycling
- free controllable recipient
- systematic use of the landfill site after fill up
- operation of the landfill mainly by public companies.


LOWER AUSTRIAN MODEL
OF WASTE MANAGEMENT
Entwurf: Pazourek,Siegl,Winkler 8/1987
Focal Point of
Research Work
C/P DISPOSAL
OF SPECIAL WASTE
THERMAL DISPOSAL
OF SPECIAL WASTE
DECENTRAL LANDFILLS
specificly selected
for Disposal of
certain types of
Special Waste
(Mono-Deposits)
RECYCLING
PLANTS
INTERMEDIATE
STORAGE
MAIN STATIONS
FOR
TRANSSHIPMENT
OF
SPECIAL WASTE
Regional Stations
for Transshipment
of problematic Waste
THERMAL
DISPOSAL
OF GARBAGE
Assistance
for Industry & Trade
LABORATORY
Collection of problematic
waste from households
Provisions by Law
Advice for
Industry & Trade
PRODUCTION
Generation of
Special Waste
TRADE
Generation of
Special Waste
CONSUMER
Generation of
Special Waste
Information
Education
Public Relation

3.3.7 The Lower Austrian Government's **way to determine locations of special waste landfills** has been as follows:

- Set up of an **inter-disciplinary working group** within the administration of the province. The team shall consist of:
  - an expert on geology
  - an expert on hydrology (hydrogeology)
  - an expert on landfill technology
  - a chemist, who is an expert on waste management
  - a chemist, who is an expert in the field of emissions/immissions
  - an expert on climatology
  - an expert on traffic and transportation
  - an expert on regional planning
  - secretarial staff.

- **Evaluation** of 18 **possible locations**[3] in 11 communities.

- **Development of a Model of Waste Management** for Lower Austria.

3.3.9 For all locations an **environmental impact study** is to be carried out parallel to the proceeding of the project.

3.3.10 The annex of this report contains tables[4] showing the **inorganic waste which can be disposed into the Special Waste Landfill**. Only solid substances with pre-scripted low solubility and water content are permitted to be built in.

3.3.11 Precise **knowledge of the origin and composition of the waste** is necessary to evaluate the destination among the different sectors of the landfill disposal.
In particular investigations have to be

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[3] The Lower Austrian Way to evaluate locations was based on extensive and careful investigations which can not be described here in detail.

[4] The tables are available in German only.

made when the delivered waste is "verunreinigt" = "impure".

3.3.12 **Operational Instructions** shall request a **detailed analysis** of the special waste, generated of a certain industry/plant. The analysis shall show all the substances contained in the waste and the dilution.

The composition of the original material has to be listed also.

Particular parameters are to be determined for the waste generating process/ plant, to characterize the specific kind of special waste. These parameters have to be checked for every delivery.

In case the parameters differ from the limits given a further full analytical investigation has to be made. According to the result, decisions are to be made whether the waste can be accepted or additional treatment is required.

3.4 Co-operation Vienna - Lower Austria

Lower Austria and Vienna are independent provinces of the Austrian Republic.

The governments of both provinces have agreed:

- to incinerate the hazardous waste of Vienna and Lower Austria at the Viennese Hazardous Waste Incineration Plant and
- to dispose of the residues of both the Viennese Hazardous and Municipal Waste Incineration Plants (e.g. slag, fly ash, filter cake) at the Lower Austrian Special Waste Long Term Landfills.

- UNIDO — General Considerations and Alternative Available Technologies (UNIDO, Chemical Industries Branch, Industrial Operations Technology Division Department of Industrial Operations, Vienna, March 1987)
- UNIDO — Expert Workshop on Hazardous Waste Management, Industrial safety and Emergency Planning, Recommendations of the Panel of Experts (UNIDO Expert Workshop, Vienna, June 1987)

Annex
p.1

Types of Inorganic Waste disposable into Long Term Landfills

31 Abfälle mineralischen Ursprungs (ohne Metallabfälle)

311 Ofenausbrüche, Hütten- und Gießereischutt

31105	Ausbruch aus Feuerungs- und Verbrennungsanlagen	A	Feuerungs- und Verbrennungsanlagen
31108	Ofenausbruch aus metallurgischen Prozessen mit produktionsspez. schädlichen Beimengungen	A	Metallerzeugung, Gießerei, Metallverarbeitung
31109	Ofenausbruch aus nicht metallurgischen Prozessen mit produktionsspez. schädlichen Beimengungen	A	Herstellung von keramischen Erzeugnissen, Herstellung und Verarbeitung von Glas, Herstellung von Carbid

312 metallurgische Schlacken, Krätzen und Stäube

31203	Schlacken aus NE-Metallschmelzen	A	NE-Metallerzeugung, -Metallgießerei
31204	Bleikrätze	A,B	Bleigießerei, Druckerei
31205	Leichtmetallkrätzen, aluminiumhaltig	A,B	Aluminiumerzeugung, Aluminiumgießerei, Aluminiumschmelzwerke
31206	Leichtmetallkrätzen, magnesiumhaltig	A,B	Magnesiumerzeugung, Magnesiumgießerei, Magnesiumschmelzwerke
31207	Schlacken aus Schmelzelektrolysen	A,B	Erzeugung von Leichtmetallen
31210	Zinkschlacke	A,B	Zinkerzeugung, Zinkgießerei
31213	Zinnaschen	A,B	Erzeugung von Zinn
31214	Bleiaschen	A,B	Erzeugung von Blei
31215	Gichtgasstäube	A,B	Eisen- und Stahlerzeugung, Eisen-, Stahl- und Tempergießerei
31217	Filterstäube, NE-metallhaltig	A,B	NE-Metallerzeugung, -Metallgießerei, Eisen- und Stahlerzeugung
31218	Elektroofenschlacken	A,B	Metallerzeugung

313 Aschen, Schlacken und Stäube aus der Verbrennung

31301	Flugaschen und Stäube	A	Feuerungsanlagen
31308	Schlacken und Aschen aus Müllverbrennungsanlagen	A	Müllverbrennungsanlagen, Sulfitablaugeverbrennung
31309	Flugaschen und Stäube aus Müllverbrennungsanlagen	A	Müllverbrennungsanlagen, Sulfitablaugeverbrennung Klärschlammverbrennungsanlagen
31310	Schlacken und Aschen aus Sonderabfallverbrennungsanlagen	A	Sonderabfallverbrennungsanlagen
31311	Flugaschen und Stäube aus Sonderabfallverbrennungsanlagen	A	Sonderabfallverbrennungsanlagen
31312	Feste salzhaltige Rückstände aus der Rauchgasreinigung	B	Müllverbrennungsanlage
31313	Feste salzhaltige Rückstände aus der Rauchgasreinigung	B	Sonderabfallverbrennungsanlagen
31314	Feste salzhaltige Rückstände aus der Rauchgasreinigung	B	Feuerungsanlagen
31315	Gipshaltige Reststoffe aus der Rauchgasentschwefelung	B	Feuerungsanlagen

314 Sonstige feste mineralische Abfälle

31419	Feinstaub aus der Schlackenaufbereitung	A	Schlackenaufbereitung
31422	Kiesabbrände	A	Chemische Industrie, Herstellung von Schwefelsäure
31423	Ölverunreinigter Boden	C	Ölunfälle
31424	Sonstige verunreinigte Böden	C	Schadensfälle
31426	Gebrauchte Kernsande	C	Gießerei
31433	Glas- und Keramikabfälle mit produktionsspez. Beimengungen		Chemische Industrie, Glasverarbeitung, Glaszubereitung, Elektrotechnik, Leuchtröhren, Lampen, Bildröhren, Wärmemeßröhrchen
31439	Mineralische Rückstände aus Gasreinigung	A	Trockene Gasreinigung

Types of Inorganic
Waste disposable
into Long Term
Landfills

31440	Strahlmittelrückstände mit anwendungsspezifischen, schädlichen Beimengungen	A,B	Mechanische Oberflächenbehandlung
31441	Bauschutt und Erdaushub mit schädlichen Beimengungen	C	Gebäude- und Anlagenabbruch, Öl- und Chemikalienschadensfälle
31445	Gipsabfälle mit produktionsspezifischen Beimengungen	B	Chemische Industrie
31447	Kieselsäure- und Quarzabfälle, mit produktionsspezifischen Beimengungen, vorwiegend anorganisch	A	Gewerbliche Wirtschaft, Keramikindustrie, Chemische Industrie, Metallurgie

316 Mineralische Abfälle

31610	Emailleschlamm, Emailleschlicker	D	Emaillierung
31619	Gichtgasschlamm	B,B	Eisen- und Stahlerzeugung, Eisen-, Stahl- und Tempergießerei
31620	Gipsschlämme mit produktionsspezifischen schädlichen Beimengungen	B	Chemische Industrie, Neutralisation, Rauchgasentschwefelung
31621	Kalkschlämme mit produktionsspezifischen schädlichen Beimengungen	B	Chemische Industrie, Neutralisation
31623	Calciumphosphatschlämme	B	Chemische Industrie
31624	Eisenoxidschlämme aus Reduktionen	B	Chemische Industrie
31626	Schlämme aus NE-Metallurgie	B,D	NE-Metallerzeugung, -Gießerei, Umschmelzwerke
31630	Bariumkarbonatschlamm	B	Härterei
31631	Bariumsulfatschlämme	B	Chemische Industrie, Papier- und Pappeerzeugung
31632	Bariumsulfatschlamm, quecksilberhaltig	B	Chemische Industrie, Erzeugung von Chlor
31633	Glasschleifschlämme mit produktionsspezifischen schädlichen Beimengungen	A	Veredelung von Glas, Glasschleiferei
31637	Phosphatierschlamm	B	Oberflächenveredelung, Phosphatierung
31638	Calciumsulfitschlamm	B	Rauchgasentschwefelung
31641	Calciumfluoridschlamm	B	Neutralisation von Flußsäure, Abgasreinigung, Aluminiumherstellung

35 Metallabfälle

353 Nichteisenmetallabfälle

35302	Bleiabfälle	A,D	Bleierzeugung, -gießerei, Druckerei, Elektrotechnik, Herstellung von Akkumulatoren und Kabeln, Bleiverarbeitung
35306	Elektronspäne	A,D	Verarbeitung von Elektron, Fahrzeugbau
35308	Magnesiumabfälle	A,B	Verarbeitung von Magnesium
35309	Zinkabfälle	A,D	Zinkerzeugung und -verarbeitung, Chemische Industrie
35315	Sonstige NE-Metallabfälle	A,D	NE-Metallgießerei, Verarbeitung von NE-Metallen
35316	Bleihaltiger Staub	A,B	Bleierzeugung, -gießerei, Druckerei, Elektrotechnik, Herstellung von Akkumulatoren und Kabeln, Bleiverarbeitung
35320	Zinkhaltiger Staub	A,D	Zinkerzeugung, -gießerei, Feuerverzinkung, Zinkverarbeitung
35321	NE-metallhaltige Stäube	A,D	NE-Metallerzeugung, -gießerei, -umschmelzwerke, -verarbeitung

355 Metallschlämme (verunreinigt)

35501	Zinkschlamm	D	Verzinkerei, Druckerei, Herstellung von Klischees
35502	Metallschleifschlamm	D	Metallbearbeitung, Oberflächenbehandlung
35503	Bleischlamm	B	Bleierzeugung und -verarbeitung, Elektrolysen
35504	Zinnschlamm	D	Zinnerzeugung, Löterei, Herstellung von Kühlern
35505	Anodenschlamm	D	Elektrolysen
35506	Sonstige Metallschlämme	B	Metallbearbeitung

Types of Inorganic
Waste disposable
into Long Term
Landfills

399 Sonstige Abfälle mineralischen Ursprungs sowie von Veredlungsprodukten

39907	Rückstände mit Elementarschwefel	A	Chemische Industrie, Herstellung von Viskose und Farbstoffen, Gasreinigung

51 Oxide, Hydroxide, Salzabfälle

511 Galvanikschlämme

51103	Chrom-(III)-haltiger Galvanikschlamm	D	Galvanikbetriebe und galvanotechnische Teilbetriebe (wie z.B. des Maschinen- und Fahrzeugbaues, der Elektrotechnik, Feinmechanik und Optik, der Herstellung von Uhren, Eisen-, Blech- und Metallwaren)
51104	Kupferhaltiger Galvanikschlamm	D	Galvanikbetriebe und galvanotechnische Teilbetriebe (wie z.B. des Maschinen- und Fahrzeugbaues, der Elektrotechnik, Feinmechanik und Optik, der Herstellung von Uhren, Eisen-, Blech- und Metallwaren)
51105	Zinkhaltiger Galvanikschlamm	D	Galvanikbetriebe und galvanotechnische Teilbetriebe (wie z.B. des Maschinen- und Fahrzeugbaues, der Elektrotechnik, Feinmechanik und Optik, der Herstellung von Uhren, Eisen-, Blech- und Metallwaren)
51106	Cadmiumhaltiger Galvanikschlamm	D	Galvanikbetriebe und galvanotechnische Teilbetriebe (wie z.B. des Maschinen- und Fahrzeugbaues, der Elektrotechnik, Feinmechanik und Optik, der Herstellung von Uhren, Eisen-, Blech- und Metallwaren)
51107	Nickelhaltiger Galvanikschlamm	D	Galvanikbetriebe und galvanotechnische Teilbetriebe (wie z.B. des Maschinen- und Fahrzeugbaues, der Elektrotechnik, Feinmechanik und Optik, der Herstellung von Uhren, Eisen-, Blech- und Metallwaren)
51108	Kobalthaltiger Galvanikschlamm	D	Galvanikbetriebe und galvanotechnische Teilbetriebe (wie z.B. des Maschinen- und Fahrzeugbaues, der Elektrotechnik, Feinmechanik und Optik, der Herstellung von Uhren, Eisen-, Blech- und Metallwaren)

513 Sonstige Oxide und Hydroxide

51301	Zinkoxid	D	Zinkerzeugung, -gießerei, Feuerverzinkerei
51302	Zinkhydroxid	D	Galvanikbetriebe, Verzinkerei, Druckerei, Herstellung von Klischees, Chemische Industrie
51304	Braunstein, Manganoxide	D	Herstellung von Batterien, Chemische Industrie
51306	Chrom-(III)-Oxid	D	Chemische Industrie, Herstellung von Pigmenten
51307	Kupferoxid	D	Chemische Industrie, Metallerzeugung
51310	Sonstige Metallhydroxidschlämme	D	Industrieabwasserreinigung, Gewerbl. Wirtschaft

515 Salzabfälle

51529	Schwermetallsulfide	D	Chemische Industrie, Erzeugung von NE-Metallen
51541	Sonstige Salze, schwer löslich	D	Chemische Industrie