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Guidelines on Remediation of Contaminated Sites

Danish Environmental Protection Agency
Danish Ministry of the Environment

Contents

1 Introduction

These guidelines serve to update, extend and supplement the General Sector Guidelines from 1992 /22/. The three sector-specific guidelines for contaminated tar/asphalt sites /25/, contaminated wood-impregnating sites /23/, and contaminated tannery sites /24/ have also been reworked and are included as Appendices 3.3-3.5 of these guidelines.

These guidelines are intended to function as a technical instruction on how to deal with a contaminated site, from the investigation phase to the remediation phase.

These Guidelines for Remediation of Contaminated Sites are among a series of guidelines which have been submitted to hearing. The others are: Guidelines for the Mapping of Contamination and Its Sources, Guidelines for Sampling and Analysis of Soil, Guidelines for the Instruction of Inhabitants in Mildly Contaminated Areas, and Guidelines for Remediation of Sites Contaminated with Oil and Petrol. The latter is Appendix 3.6 in the present guidelines.

The directions given in these guidelines are not binding, but should be seen as the foundation on which the authorities, companies and consultants can base a more equal treatment of cases concerning contamination.

However, the authorities should always take the directions given in the guidelines as their point of departure when dealing with cases concerning contamination. The directions given in these guidelines should always be adapted so as to meet the requirements of the existing legislation. In relation to the current legislation, this means that the guidelines cannot be applied directly to the restoration principle of the Environmental Protection Act or to the de-listing situations in the Contaminated Sites Act.

This is because a complete removal of contaminated soil from the site is not always necessary in situations where land use and management of the contaminated soil is regulated administratively.

2 Strategy

2.1 Division of efforts into phases

The choice of strategy is determined by the objectives of the investigations. An investigation may be carried out in connection to the purchase/sale of property, for the purpose of listing the site as contaminated or for the purpose of remediating the site. These guidelines deal primarily with investigations carried out in connection with risk assessments of human health and the environment, and with possible subsequent remediation with regard to the established risk.

When choosing which strategy to apply, it is appropriate to divide the process into the following stages:

- The initial survey phase
- The investigation phase
- The remediation phase
- The operation and evaluation phase

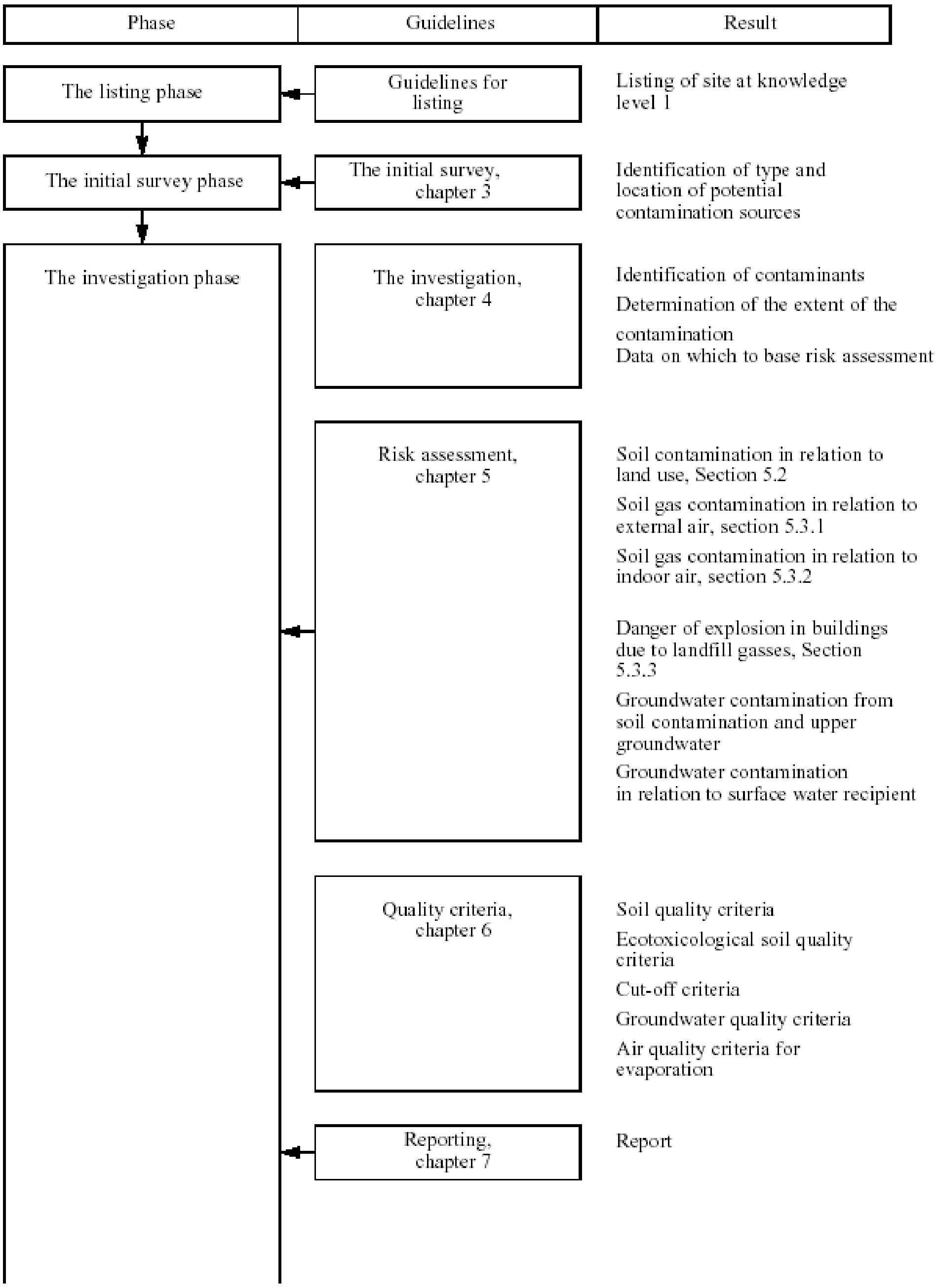
The division into phases is a good way in which to divide the work that is to be done. After each phase, the need for further measures is assessed. The division thus aims at optimising the use of information obtained in one phase for the planning of subsequent actions.

The reporting of activities that have been implemented does not necessarily follow the division of phases. Several phases can be reported at the same time (e.g. the initial survey phase and the investigation phase), and several reports can be made in one phase (e.g. following both the preliminary and the supplemental portion of the investigation phase).

Much is already known about investigations and remediation within certain sectors. This entails that, in some sites, phases can be ‘grouped’. Furthermore, mapping information /1/ or listing investigations are already available at many sites, thus overlapping the initial survey phase and the investigation phase.

So as to implement investigations and remediation as rationally and appropriately as possible, the strategy, and therefore the choice of phase division, is based on the available knowledge regarding the site. In some instances, it can be practical to implement investigations and remediation in accordance with the procedures applied in ordinary building and construction projects /2/. In the following, the contents of the four phases will be described briefly.

The connection between the four phases and the sections in these guidelines are shown in figure 2.1.



Phase	Guidelines	Result
The listing phase	Guidelines for listing	Listing of site at knowledge level 1
The initial survey phase	The initial survey, chapter 3	Identification of type and location of potential contamination sources
The investigation phase	The investigation, chapter 4	Identification of contaminants Determination of the extent of the contamination Data on which to base risk assessment
	Risk assessment, chapter 5	Soil contamination in relation to land use, Section 5.2 Soil gas contamination in relation to external air, section 5.3.1 Soil gas contamination in relation to indoor air, section 5.3.2 Danger of explosion in buildings due to landfill gasses, Section 5.3.3 Groundwater contamination from soil contamination and upper groundwater Groundwater contamination in relation to surface water recipient
	Quality criteria, chapter 6	Soil quality criteria Ecotoxicological soil quality criteria Cut-off criteria Groundwater quality criteria Air quality criteria for evaporation
	Reporting, chapter 7	Report

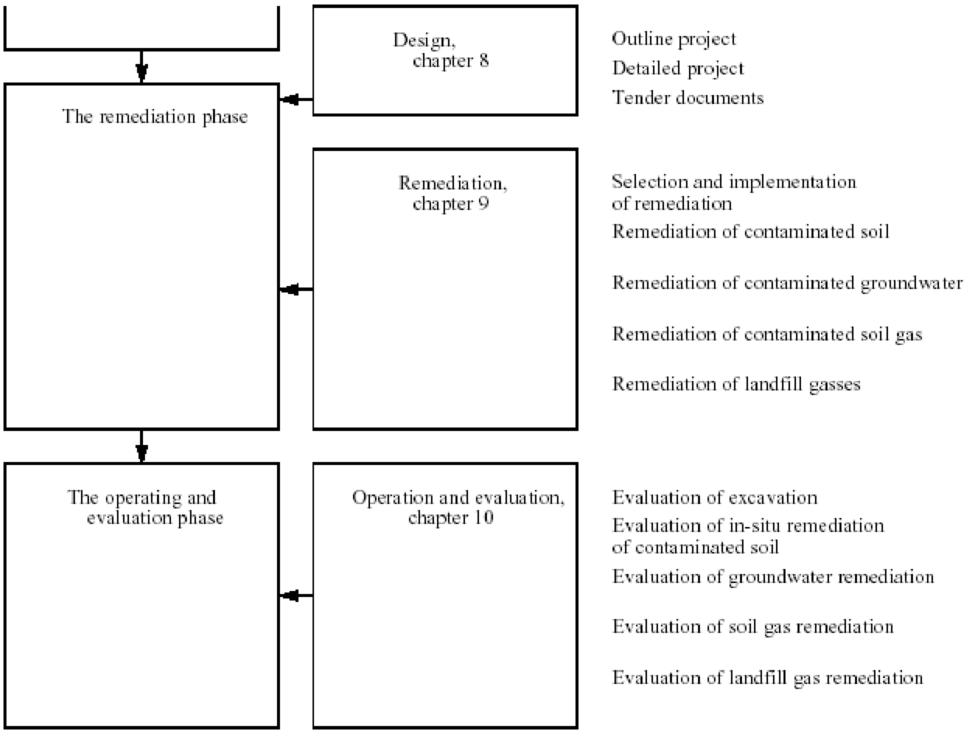


Fig. 2.1
Division of activities into phases.

2.2 The initial survey

The objective of this phase is to obtain the best possible basis for implementing the investigation of a contaminated site.

In the initial survey, information about the site which can support further measures should be collected. This should include information from existing charts and maps. In this phase, a historical review of the site should be prepared with the purpose of advancing a hypothesis regarding which contaminants may be present and where contamination can be expected.

2.3 The investigation phase

2.3.1 Objective

Investigations of contaminated sites typically aim at describing:

- The collection of representative data to be used in risk assessment.
- The extent and intensity of soil contamination.
- Indoor air problems in buildings due to evaporation of hazardous substances.
- Risk of explosions in buildings and installations due to landfill gas.
- Spreading of contaminants in superficial and deeper groundwater and possibly to a surface water recipient.

The investigation phase comprises:

- Investigations

- Risk assessment
- Reports
- Outline project

Investigation techniques include boring, collection of soil, water and air samples, characterisation of samples and chemical analysis of samples.

Sampling should be planned in accordance with the investigation’s objective to ensure that the number and the selection of samples ensure that the objective can be met. This has been described in further detail in Guidelines for Sampling and Soil Analysis /3/.

If it is possible to foresee a remediation project before the investigation phase is put into action, then the investigation should be planned so as to be aimed towards this project.

Field and laboratory investigations should be planned so as to be aimed at collecting data which can be used in risk assessments of groundwater, land use and surface water recipients.

Investigations and risk assessments should always be reported. If investigations are carried out in several stages, such as preliminary and supplemental investigations, these should be reported independently of one another. However, all results should be taken into account in the overall risk assessment.

In cases where remediation is necessary, the investigation phase should be concluded by drawing up an outline project. Several different proposals for remediation techniques are typically discussed with the authorities before the outline project is drawn up.

2.3.2 Preliminary investigations

The objective of preliminary investigations is to test the hypothesis put forward in the initial survey, and to obtain an impression of how contaminated the site is.

The strategy for preliminary investigations should be based on information that has been gathered in the initial survey.

Soil borings should typically be placed at locations where contaminants are likely to be found, and should determine the extent of previously known contamination. Soil borings should be placed in such a manner as to cover areas with contamination-sensitive land use as well as other areas, where contamination may be found.

Placement of number of soil borings are dealt with in more detail in Chapter 4 and in Guidelines for Sampling and Analysis of Soil /3/.

When preparing for water sample collection, it is advisable to construct wells in the source area or immediately down gradient in order to determine the contamination strength of the source. When estimating the strength of the source, the distance between the well and the source of contamination - and thus the preceding dilution - should be taken into consideration.

Sometimes it is possible to take water samples in the sand fill around a tank or pipeline so as to establish the strength of the source.

The final sampling strategy is based on data collected in the initial survey and an analysis programme should be prepared.

The analysis programme in a preliminary investigation is typically quite general in nature. In supplemental investigations, when the problems have been identified, a more narrowly targeted programme can be applied.

Preliminary investigations should be concluded with an assessment of whether sufficient evidence has been established which illustrates the site's contamination and which enables reliable and representative risk assessment.

Should the preliminary investigations indicate that the site is so contaminated that more detailed information is required, a proposal for supplemental investigations should be prepared.

2.3.3 Supplemental investigations

The objective of supplemental investigations is to clarify problems which have been established in preliminary investigations.
The investigations should typically be performed in order to:

- Give a more detailed description of type, severity and extent of contamination in soil or groundwater near the surface.
- Clarify land use possibilities.
- Assess the risk of indoor air problems, including the risks from landfill gasses, in existing buildings and installations on and near the site (contaminants transported via the sewer system or groundwater).
- Assess the possible effect of contaminants on deeper groundwater aquifers or nearby surface water recipients.
- Prepare an outline project, if needed.

If remediation is deemed necessary, supplemental investigations should lead to one or more outline projects. Outline projects should consist of an overall description of the fundamental technical solutions and include a rough estimate of the cost of such a solution and a time table.

2.3.4 Risk assessment

Risk assessments are assessments of the environmental and health consequences of a given contamination. The objective of risk assessments is to establish the need for remediation.

Risk assessments are based on specific situations and information on the contaminants, contaminant transport and exposure pathways, and the target group in question in the given situation. The risk related to land use, to evaporation and to groundwater should be assessed independently of one another.

Criteria for soil quality have been established to be used in risk assessments in relation to land use. Sites where the soil fully lives up to these criteria can be used without restriction for all purposes including those that are highly sensitive to contamination. Furthermore, a cut-off criteria has been established for several contaminants. These criteria state the level at which it is necessary to prevent all contact with the soil. Principles for remediation (such as clean-up depths) and principles for assessment of soil contamination are stated in Section 9.2. It should be noted that compliance with soil quality criteria does not necessarily ensure compliance with evaporation and groundwater criteria.

At contaminated sites, indoor air in buildings as well as outdoor air may be unacceptably affected by underlying contamination of soil or groundwater. The effect on the indoor and outdoor air should be assessed in stages by various methods that have been constructed. In several stages, theoretical calculation models are included. Risk assessment should be based on evaporation to the overlying air and must not exceed the acceptable contamination contribution.

The risk of methane gas explosions in buildings on or in the immediate proximity of landfill sites can also be assessed in stages.

Risk assessment of groundwater should be used to assess whether contamination of either soil or upper groundwater aquifer contributes unacceptably to the contamination of groundwater resources. Groundwater quality criteria have been established for use in risk assessments. Risk assessments should be based on the aquifer complying with groundwater criteria at all points. Risk assessment can be carried out in stages, starting with a simple assessment. If this assessment does not provide enough evidence of the lack of risk, more complicated calculations should be carried out. Sorption, dispersion and degradation of the contaminants should be taken into account in these calculations. Furthermore, the assessment of the effect on the groundwater should be used to perform a risk assessment for surface water recipients.

If the risk assessment establishes a risk to human health or the environment, residents on or in the proximity of the site should be advised as to how to act until remediation can be implemented.

2.4 The remediation phase

The objective of the remediation phase is to plan in detail and implement the required remediation. The aim of remediation is to remove contaminants, limit exposure or prevent the spread of contaminants to soil, water or air.

Remediation can be extremely varied. It can involve a simple excavation of soil near the surface, a long-term pump-and-treat technique for contaminated groundwater, or a complicated in-situ techniques.

In the course of detailed planning, it is often necessary to perform supplemental investigations that specifically aim at the chosen remediation technique. Investigations could, for example, include detailed mapping of the extent of contamination or a pump test for designing soil vapour extraction.

If the concentration of the contaminant is below cut-off criteria, the precautions stipulated in Guidelines Regarding Advice of Residents in Mildly Contaminated Areas /4/ can be carried out.

2.5 Operation and evaluation phase

The objective of the operation and evaluation phase is to check the effect of remediation.

Before the operation and evaluation phase is put into action, procedures for evaluating measured parameters should be prepared. These should include alarm values with a view to adapting the remediation and stop criteria with a view to stopping the operation and evaluation phase. Procedures should also describe the frequency and form of reports, in which operation and evaluation measures should continuously be assessed to ensure that the required environmental effect is achieved.

2.6 Strategy for distribution of information

Depending on the extent of the actions, residents who are affected by these should be informed of what is being done. This kind of information should be an integral part of the entire process. A high level of information can ensure that the planned activities can proceed as smoothly as possible.

The problem holder is responsible for providing residents with the necessary information. This can be done by sending letters of information. Residents in the area are best informed by holding information meetings supplemented by the letters of information.

Residents should receive information regarding the planned activities before the first measures are put into action. This information should be followed up throughout the entire process, informing the involved parties about planned actions, results, conclusions and consequences. Furthermore, the information should contain a time table for the entire operation, including plans for further information.

3 Initial survey

3.1 Introduction

During the initial survey, all information on the relevant site which is immediately available, should be obtained. The primary objective at the majority of industrial sites is to identify both the nature and physical location of the potential sources of contamination.

Information collected in the initial survey provides the foundation for the rest of the course of the investigation. Therefore, the initial survey should be executed with the greatest of care.

The initial survey comprises:

- Collection of historical data on use of the site, including information on possible listing of the site, cf. Guidelines on Mapping /1/.
- Acquisition of geological and hydrological data for the area.
- A site visit.
- An assessment of the data collected and a hypothesis on possible contamination.

3.2 Previous and current land use

As far as possible, the previous and current land use of the site should be described through acquisition of data on:

- The precise location and extent of the site. It is important here to note that a site can previously have extended further than the current limits, if the area has been parcelled out.
- All building activity and possible alterations to the terrain.
- The types of enterprises and possible other land use of the site in chronological order.
- All potential contaminating activities on the site are to be determined from information on production, including which equipment and processes have been used. It may be relevant to obtain information from different periods of the enterprise’s operations, since production methods may have changed.

There are a large number of sources from which information may be obtained. Table 3.1 lists a number of useful sources. Based on previous experience, the sources are divided into primary and secondary sources. The most important information is to be found in primary sources. If the primary source is thought to be inadequate, supplementary information can be sought in the secondary sources. This division is a recommendation only, since the need for information is naturally determined by the individual site. Furthermore, refer to ‘Guidelines on Mapping Soil Contamination and Sources’ /1/.

Table 3.1
Overview of sources of information on previous and current land use of sites.

Primary sources
Local authority records Local historical records Background material on equipment and processes Interviews and investigations Company records Land Registry Office The police and fire departments
Secondary sources
Register of Companies etc. The Working Environment Service The Royal Library The Danish National Business Archive in Aarhus National Survey and Cadastre

Below is a short introduction to the sources mentioned in Table 3.1.

Local authority records
Local authorities maintain records of construction activities, including construction of sewage systems. They also keep records on environmental approvals and inspections, including contamination which has been ascertained, and they have records of underground oil and chemical storage tanks, as well as enterprises which generate chemical waste.

Some local authorities keep all the above information on a specific site in a single file, but filing practises vary from local authority to local authority.

Environmental permits contain descriptions of production processes, measures to limit contamination, and waste products and their disposal. Environmental permits cover the period after the Environmental Protection Act entered into force in 1974. Certain types of enterprises with regional impact are listed in the Act. These enterprise are supervised by the regional authorities.

Local historical records
At local historical archives, old maps, telephone directories, and information booklets may be available. There may also be collections of relevant photographs and newspaper clippings. Furthermore, personnel at these archives often have extensive local knowledge.

Background material on equipment and processes
General knowledge of production techniques, processes, raw materials, and chemicals etc. can be found in specialist literature and at sector organisations.

Materials which are specially relevant to the soil and groundwater have been prepared for a number of types of enterprise/sector. Besides descriptions of the enterprises'/sectors' production and potential sources of contamination, including parameters for analyses, several references include descriptions of previous experience from earlier investigations, as well as soil and groundwater contamination which has been ascertained at the site. References to this material are included in Appendix 3.1.

Interviews and site visits
Interviews with previous and current employees can support and supplement information from records and literature. If possible, the collection of data should therefore include interviews.

A site visit should be carried out. During the walk through, information collected from records and archives should be checked against current conditions. The location of existing buildings and installations should be recorded and visible signs of soil contamination should be noted.

Moreover, it may be important to note information regarding access to the site for use in planning boring activities, etc.

Appendix 3.2 provides a checklist which can be used for the site visit.

Company records
The enterprise may maintain relevant information. Examples include: statements or data booklets regarding quantities of raw materials used and products produced, old photographs, or drawings.

Land Registry Office
Information on previous owners of individual sites is registered here. Information can be obtained from the Land Registry Office in the individual local authority by checking the declarations for the relevant cadastre and the appendices to these.

Police and fire departments
Information on previous stores of flammable and explosive substances may be kept by the local police or fire department. These authorities may also contribute with chronological information on actions which may have environmental significance, e.g. fire or other accidents such as spills, leaks, or overflows from tanks. In some cases, they may have information from the local authority records from before 1970.

Register of Companies etc.
More detailed information companies registered as limited can be obtained from the Register of Companies, Kompass Danmark, or Greens Danske Fonds og Aktier (Greens Danish Funds and Shares). These documents often include the company’s primary activities.

The National Working Environment Authority
Information on chemicals and accidents can be obtained from the National Working Environment Authority. To access historical archives the name of the enterprise is used, rather than address or cadastre numbers.

The Royal Library
A limited number of aerial photographs from before 1945 are kept at the Royal Library.

Aerial photographs can be useful in obtaining an impression of land use at the site. Tanks, barrel/drum stores, and waste can be localised from aerial photographs. The library also has a large collection of maps.

The Danish National Business Archive
'The Danish Tariff Association's Archive' (*Dansk Tarifforenings Arkiv),* which is at the Danish National Business Archive in Aarhus contains a lot of relevant information based on insurance companies' inspections of all larger enterprises (app. 50,000) in the period 1896-1982.

The register can be inspected at the Danish National Business Archive, while access to the reports themselves requires permission from the owner, the Danish Assurance Association (*Dansk Skadesforening*).

National Survey and Cadastre
Aerial photographs from 1945 onwards are available from the National Survey and Cadastre. Use of aerial photographs is described under 'The Royal Library'.

Finally, the Danish National Museum has information on registration of industry.

3.3 Surface water recipients and soil and groundwater conditions

When obtaining information on geological and hydrogeological conditions in the area, a preliminary vulnerability assessment may be made.

In addition, an overview should be obtained of water abstraction, groundwater flows, and surface water recipients in the area.

Data for descriptions of soil and groundwater conditions, as well as surface water recipients is obtained from:

- Topographical maps (scale 1:25,000)
- Geologic basic data maps (with Danish cyclogram symbols)
- Maps of the groundwater's potentiometric surface
- Water abstraction plans
- Water supply plans
- Geological literature
- Other investigations in the area

Information on the potentiometric surface, water abstraction, and groundwater quality can be obtained from the regional authorities.

The regional authorities, and to a large extent the Geological Survey of Denmark and Greenland (GEUS) and local authorities, can contribute with information on the location of borings and wells in proximity of the site. In addition to the location of borings, information on the geological strata and the depth of the groundwater is available.

3.4 Reporting

The results of the initial survey should be reported in such a way that the data collected, cf. Sections 3.2 and 3.3, is presented clearly and comprehensibly. The results should be assessed and related to the original hypothesis for possible contaminating activities at the site.

Sources of information should be stated, cf. Table 3.1, and possible ‘holes’ in the (historical) information collected should be considered.

There are advantages in reports comprised of a data sheet with relevant maps as appendices. The data sheet states information in table and in the form of key words. On one or more maps, relevant characteristics of buildings or production are shown, including where potential contaminating activities have taken place/take place.

4 Site investigations

4.1 Introduction

The scope of the investigation of contamination, and thus the collection of data, depends entirely on the result of the initial survey as well as the requirements for the data which are to be used in connection with the subsequent risk assessment. Part of the information from the Guidelines on sampling and analysis /3/ have been included in this chapter. This chapter includes the following main sections:

- Sampling soil and water
- Sampling air
- Methods of analysis
- Collection of data on buildings
- Geology, hydrogeology, and hydrology

4.2 Sampling soil and water

Soil and water sampling is carried out in the investigation phase, usually from borings and monitor wells. The objective of sampling is to describe the contamination, the soil, and the groundwater to such an extent that risk analyses may be carried out, and so that there is an adequate foundation for planning and executing necessary remediation.

4.2.1 Location of soil borings

Soil borings are usually located as follows:

- in areas where contamination is expected (hot spots) on the basis of knowledge of the location of previous and existing production plants, and on the basis of other land use.
- near the boundaries of already known contaminated areas for the purpose of determining the extent of contamination.
- in areas which have, or which are suspected to have, a contamination-sensitive land use. Such areas are covered with a relatively high intensity of borings as described below.
- at the rest of the site, since it is not always possible to localise all hot spots during the initial survey, and as spills are often spread over large parts of a site.

In order to reveal unknown contamination and to achieve the best statistical coverage of the site, it may be beneficial to locate borings according to specific rules. In such cases, sample fields/grids/nets can be defined. Table 4.1 describes the number of sample points which are necessary to localise unknown hot spots.

Table 4.1
Number of sample points which are necessary to locate a hot spot of given size, with given certainty. in a 400 m^2 area.

Diameter of hot spot (m)	% contam. site %	Probability			
		50%*	90%*	95%*	99%*
10	20	2	5	6	7
7	10	5	10	12	15
5	5	10	20	24	29
3	2	28	54	65	81
2	0.8	62	122	147	183
1	0.2	249	488	589	731

* Probability of localising a hot spot

For example, the table shows that if 24 sample points (borings) are placed on a 400 m^2 site, there is a 95% probability of finding a contamination with a diameter of 5 m.

It is therefore financially unrealistic to localise unknown, smaller hot spots with high probability. Normally, investigations will be based on known sources of contamination and how contamination has dispersed from these. When using only a few borings, it is not possible to assume that all small hot spots have been localised, nor that the contamination has been fully delineated. It is important to stress that each boring only represents a point measurement. Therefore, the results from such point measurements should be interpreted with care during the assessment of a contamination.

The location and number of borings is described in more detail in the Guidelines on sampling and analysis of soil /3/, where there are examples of sampling strategies, and suggested sample densities if the source of contamination has not been localised, or if there are special objectives. As an absolute minimum, representative samples should be taken which correspond to a rough screening in Table 4.1.

4.2.2 Execution of borings/excavations

The objective of executing borings is to obtain representative samples to determine geological and contamination conditions horizontally and at depth. Appendix 4.1 contains more detailed guidelines on executing borings.

When carrying out soil sampling very close to the surface, borings may be replaced by excavations. Excavations are usually carried out with a trench digger. Only in rare circumstances will digging by hand be advantageous. Excavating by machine provides a good overview of the soil layers, and how the contamination varies along the face of the excavation. Excavations are particularly advantageous in cases where the contamination is distributed unevenly, for example landfills, or if an investigation of other geological conditions is desired. It is important to note that excavations are only advantageous if, prior to excavating, permission is obtained to refill the trench with the excavated soil, even if it is contaminated.

Localisation borings are shallow investigative borings of up to 3-4 m in depth. Localisation borings are made in order to describe and delineate a contamination in the upper soil layers and/or groundwater aquifers close to the surface.

Often, very shallow borings of up to 1 m are made for contamination such as metals close to the surface. These borings can usually be back filled with excavated material. If a boring is deeper than a maximum of 3-4 m, casing should be used to avoid possible cross contamination. The boring method usually used is dry rotation boring. Methods of boring and screen installation are described in detail in Appendix 4.1.

Investigation borings are borings of more than about 4-5 m where, in addition to describing and delineating a contamination in the upper, and possibly deeper soil layers, the objective is also to obtain information on deeper groundwater. Borings with this depth should always be carried out using casing in order to ensure representative soil samples and in order to prevent cross contamination when boring through several water-bearing strata. The method of boring most commonly used is dry rotation soil boring with casing. Methods of boring and screen installation are described in detail in Appendix 4.1.

Wells are installed to investigate deeper aquifers, to monitor groundwater or to remediate groundwater by pumping. Different methods of boring are described in Appendix 4.1. In many cases, wells are screened in upper aquifers.

4.2.3 Soil sampling

During boring, a set of two soil samples is usually collected at each depth. One sample is used for characterisation, including field measurements, cf. Table 4.5, and for geological descriptions. The second sample is used for chemical analyses. Sample sets are usually collected every 0.5 m. As a minimum, however, one set is collected per soil layer.

Methods of sampling, packing, handling, and storage should be adapted to the type of contamination. It is vital that guidelines are followed, especially for volatile contamination, as investigations will otherwise lose their value. Guidelines on soil sampling are described in more detail in Appendix 4.2 and in the Guidelines on sampling and analysis of soil /3/.

4.2.4 Water sampling

The objective of sampling is to obtain a water sample from the well which is representative of the aquifer with regard to the parameters to be investigated.

Water samples are usually collected from screens in wells. In screened localisation borings, it is normal to install screens only in the upper saturated zone. Placement of screens in investigation borings depends on the hydrogeological conditions and the specific objectives of the borings, cf. Appendix 4.1.

Sampling comprises three phases:

- Purging
- Sampling
- Sample storage

There is a distinction between well development and purging. Immediately after a well is completed, well development should be carried out by pumping in order to achieve the best possible well efficiency. Well development can be conducted as part of stepwise pump test, cf. Section 4.2.5.

The groundwater in screened investigation borings is in contact with the air. This means that the temperature, the oxygen content and the carbon dioxide content of the groundwater in the well and its immediate vicinity can be significantly different from conditions within the aquifer. These differences can also mean that the content of contaminants is not the same in the aquifer and the well, due to chemical and biological activity. Furthermore, there is a risk that volatile compounds will have evaporated from the water in the well.

In order to ensure collection of a sample which represents the groundwater aquifer as well as possible, purging should be carried out prior to sampling. Different types of pumps are used for purging and water sampling, depending on the nature of the boring and the hydraulic conditions. The various types of pumps and the approach to purging are described in more detail in Appendix 4.3.

The objective of sampling is to obtain a water sample from the aquifer via the well. During this phase, there are three factors in particular which are significant:

- The equipment should not contaminate the sample.
- The equipment should not be made of materials which ad/absorb substances.
- The method should not bias the contaminant content of the sample.

Water sampling is described in more detail in Appendix 4.3. Appendix 4.10 provides examples of forms which can be used for water sampling. Some parameters should be measured on site (O_2, CO_2, E_k (redox potential) and pH), as these may change if the sample is transported.

The packing which is used to store the sample during transport to the laboratory should ensure that the samples change as little as possible. The sample container should be cleaned by the laboratory before delivery to the site. Water samples which are to be analysed for organic parameters should be stored in glass flasks with tight lids. Samples which are to be analysed for inorganic parameters, e.g. heavy metals, are often stored in plastic bottles. For certain parameters, the laboratory will provide sample containers which are specially cleaned, or which contain a preservative.

Water samples should be stored in the dark and kept cool (4^0C). The time from sampling to analysis should be kept to a minimum. As far as possible, samples should be delivered to the laboratory on the same day they are collected. If this is not possible, it should be noted on the analysis form.

4.2.5 Execution of pump tests

Pump tests are carried out to determine the physical and geometric properties of the groundwater aquifer. Pump tests provide information on the water level as a function of time, yield, and distance.

A step-test is carried out to determine the characteristics of the well. Well loss arising from the screen and the total well efficiency is determined. A step test is therefore primarily performed to demonstrate ‘how good the well is’, i.e.

how much the well can produce in relation to the potential yield of the aquifer.

Immediately following the well completion, development of the well should be executed, possibly as a beginning of a step-test. Thus, in addition to developing the well, information is also obtained on how good the well is and possibly on the hydraulic parameters of the aquifer.

Pumping is usually carried out as a minimum in a three-step test with variable yields, according to the regulations in the Statutory Order on Execution of Groundwater Wells /6/. Regular yield measurements should be taken at all steps. Head measurements should be more frequent at the start than at the end of the test, for example as described in Appendix 4.9. After the final step, head recovery in the well is measured with the same time intervals. From this data, well efficiency and well loss can be calculated.

Pump tests at constant capacity are carried out in order to determine the hydraulic parameters of the groundwater and the horizontal limits of the aquifer. In principle, an investigation of a groundwater aquifer with a pump test is carried out by pumping at a constant capacity from an individual well while observing the drawdown in selected observation wells (well test using drawdown data). After pumping is stopped, recovery in the observation wells is measured to obtain control measurements (well test using recovery data).

The drawdown/recovery which is observed is treated as a function of time and distance, and is interpreted to determine the hydraulic properties of the aquifer; transmissivity, storativity/specific yield, and leakage. Furthermore, information may be obtained on the aquifer’s boundary conditions, recharging boundaries (e.g. watercourse) or discharging boundaries (e.g. clay aquitard). Information can also be obtained on the anisotropy of the aquifer.

Once interpretation is completed, it is in principle possible to calculate future drawdown for any given yield, possibly as an aid in designing groundwater remediation using the pump-and-treat technique. A more detailed description of pump tests is included in Appendix 4.9.

4.3 Air sampling

4.3.1 Measurements of organic and inorganic gases in soil gas

In the unsaturated zone, contaminants are distributed in three phases: absorbed on the soil, dissolved in the soil water, and as gas dissolved in the soil gas (with heavy contamination, there can also be contamination components in a separate phase). The distribution between the three phases depends on the physical/chemical properties of the contaminants.

A greater proportion of soil contamination will be gaseous for volatile compounds. As a result, it can often be advantageous to carry out soil gas measurements when investigating the contamination, taking into account that the soil must have a certain conductivity.

Soil gas measurements are typically used for highly volatile hydrocarbon contaminants, such as benzene, toluene, and xylenes, or for chlorinated solvents. However, soil gas measurements can also be used for other contaminant components, such as naphthalene and hydrocyanides.

Soil gas measurements are particularly useful for:

- Risk assessment of very sensitive area land use, including contamination of indoor air.
- Preliminary contamination investigations where contamination with volatile compounds is suspected.
- Mapping the extent of soil contamination or groundwater contamination near the surface.
- Localising point sources of identified soil or groundwater contamination, for example at landfills.

Soil gas measurements are carried out by ramming a probe down to a given depth in the unsaturated zone. Soil gas investigations can also be carried out from a vehicle specially equipped with sampling and analysis equipment. The typical sampling depth is 1-5 m, depending on the objectives, geology, and expected contamination. Soil gas is pumped up from the probe and is collected for subsequent analysis in, for example, tedlar bags or on adsorption tubes. The method of collection depends on the method of analysis chosen. When assessing indoor air risks, soil gas measurements can also be carried out in the capillary-breaking layer under the floor of a building. Methods of measurement are described in more detail in Appendix 4.4.

The subsequent analyses can either be conducted as field measurements using PID instruments (photo-ionisation detectors), portable gas chromatographs, etc. or they may be conducted in the laboratory. The advantage of field measurements is that results are available immediately after samples are collected and thus ongoing adjustments can be made to the investigation. However, laboratory analyses have the advantages of less uncertainty and lower detection limits.

Selection of the method of analysis depends on the application of the soil gas measurements. For delineation of a known contamination, where rapid results may be important, it is most appropriate to carry out field measurements, while for preliminary contamination investigations or for assessment of risks to the indoor air, analyses in the laboratory may be more appropriate.

4.3.2 Landfill gas

The primary objective of gas investigations at landfills and tips where biodegradable waste has been deposited is to obtain an assessment of whether the methane gas which percolates up can lead to a risk of explosions in buildings on the site or in the immediate vicinity.

Appendix 4.5 provides brief information on landfill gas. Investigations prior to a risk assessment includes:

- Data collection of the conditions at the landfill, including local geologic and groundwater conditions.
- Determination of the gas-generating areas, including the pressure conditions in the area.
- Data collection of building conditions, including underground cables and pipes.

To start with, accessible (historical) knowledge of the parameters which have an influence on gas generation and gas transmission out of the site is acquired.

Table 4.2 provides a checklist which can be used in collecting this information.

Table 4.2
Checklist of conditions at a landfill/deposit

- Type of waste deposited
- Amount of waste deposited/fill-layer thickness
- Period over which waste was deposited
- Local geology
- Groundwater conditions
- The top layer at the landfill/deposit

In order to identify the areas generating gas, including identifying the strength of the source, soil gas measurements must be taken at the site. Appendix 4.6 describes guidelines for conducting soil gas measurements of landfill gas.

Seasonal variations, meteorological conditions, etc. affect results significantly. It is therefore advisable to carry out soil gas measurements of landfill gas three times over a year before making final decisions. Endeavours should be made to carry out measurements under situations where the surface is saturated with water, or covered with snow, when there is a change from high to low pressure, and at high ground temperatures.

As gas may be transmitted horizontally in porous soils, soil gas measurements should cover both the landfill site and any nearby critical areas. The potential critical distance depends on the local geology. Illustrative critical distances are provided in Table 4.3.

Table 4.3
Potential critical distances from landfills, cf. Figure 4.1

Moraine clay	10 m
Fine sand	25 m
Coarse sand	250 m

It is advantageous to locate soil gas measuring points in a grid (e.g. 50 x 50 m). A diagram of suggested measuring points for an investigation is shown in Figure 4.1. Section 5.3.3, reviews results analysis, including assessment of the actual critical distance.

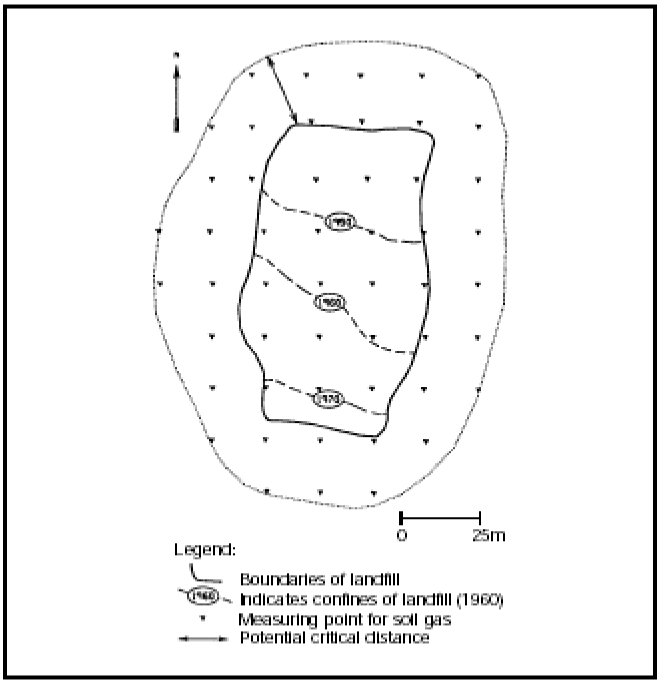


Figure 4.1
Illustration of measuring points for investigations of landfill gas

Figure 4.1 shows the boundaries of an old quarry which has now been landfilled. The quarry was landfilled from north to south from about 1940 to 1972. With the aid of aerial photographs, the progress of the landfilling can be identified at various points in time. The local geology comprises primarily fine sand. Table 4.3 below shows the potential risk of critical gas dispersion to a distance of 25 m from the edge of the landfill.

In order to investigate whether landfill gas is still being generated at the site, soil gas measurements are taken at a large number of soil gas measuring points. The measuring points are located in a network of about 25m x 25 m.

In order to be able to assess the risk of gas penetrating a building located on or near to the landfill, information should be collected on how the building is constructed. Furthermore, information should be obtained on underground cables and pipes, as well as possible other underground installations where there is a risk of spreading the gas. Table 4.4 provides a checklist for this purpose.

Table 4.4
Checklist of construction conditions of a building.

- Underground cables etc.
- Other underground installations
- Construction of foundations and joists
- Construction of flooring
- Recording of cracks in floors
- Recording of pipe lead ins (entry points)
- Ventilation (e.g. ceiling height, air renewal)
- Distance

Information on the construction of buildings can be collected after the gas-generating area has been identified and the actual critical distance for surrounding buildings has been determined, cf. Figure 5.8 in Section 5.3.3.

4.3.3 Indoor air

In special cases, it may be necessary to conduct indoor air investigations in existing buildings in connection with health assessments of the indoor air. Methods and procedures for measuring air quality are described in guidelines from the National Housing and Building Agency on measuring substances in indoor air arising from soil contamination /7/. As a minimum, measurements should be made in three areas:

- The place in the building where the greatest concentration of contamination from the soil is expected.
- The place in the building where there is least risk of finding contamination from the soil. This place should be in frequent use by the residents.
- Some typical concentrations from outdoor air.

The measuring sites and the number of series of measurements are set on the basis of the above. Furthermore, there are a number of factors which can have a crucial effect on the results of the measurements, e.g. barometric pressure, wind velocity, air pressure in the building, indoor activities and building materials. The significance of individual factors is described in the guideline from the National Housing and Building Agency /7/, which includes recommendations on what should be done.

4.4 Methods of analysis

The objective of analysing soil and water samples is to identify the degree of contamination in the area being investigated.

There are various methods of analysing soil and water samples. The methods of analysis vary in price, speed of analysis, the type of substance to be analysed, detection limits, accuracy in quantifying the content, and precision.

Many screenings or on-site methods are less accurate than other analyses, but they provide results for more substances in the same analysis. Some on-site methods do not allow direct quantification of the content of individual contaminants, but provide more qualitative indications of whether there is a high or low content of specific types of substances.

When investigating unknown contamination, there is a need to analyse for a broad spectrum of substances. Here, it can be an advantage to utilise these less accurate, but broader, methods of analysis in order to achieve a wider description of the contamination. The number of analyses is more significant in localising or delineating a contamination with certainty rather than the accuracy of an individual analysis.

4.4.1 Field methods

The various types of field methods are listed in Table 4.5. The methods range from non-specific to substance-specific.

Table 4.5
Field methods and analysis parameters /3/.

Field method	Analysis parameters
Non-specific methods	
Visual assessment	Oil, tar, slag, cyanides
FID	Chlorinated solvents, petrol, phenols, oil, (tar)
PID	Chlorinated solvents, petrol, phenols, oil, (tar)
Contamination-specific methods	
Immunoassay	Oil, petrol, PAHs, polychlorinated phenols, TNT
Infrared spectroscopy	Oil, petrol, chlorinated solvents
Test tube	Petrol, water miscible solvents, chlorinated solvents, cyanides
Colour reaction	Petrol, oil, polychlorinated phenols, (metals)
Electrochemical potential*	Oil
Fluorescent measurements (UV)*	Oil, tar
Optic fibre techniques*	Oil, petrol, chlorinated solvents
Substance-specific methods	
GC/PID/FID/ECD, headspace over soil, extraction of soil	Petrol, oil, chlorinated solvents, water miscible solvents
EDXRF	Heavy metals
Thin layer chromatography*	Oil, tar, pesticides, polychlorinated phenols

*Methods under development

The most commonly used non-specific method is measurement of volatile organic substances using a photo-ionisation detector (PID) or a flame-ionisation detector (FID). Measurement with a PID is described in more detail in Guidelines on sampling and analysis of soil /3/. The FID functions on the same principle as the PID, but it has a different type of detector which covers a larger area than the PID.
This method measures the air immediately above the soil sample. Therefore, only volatile substances in the gaseous phase are measured. Portable gas chromatographs are available with different detectors. The most common is the photo-ionisation detector. Besides this, there is the flame-ionisation detector and the electron capture detector (ECD). Gas chromatographs are substance specific, i.e. individual substances can be detected using this method.

X-ray fluorescent measurements for heavy metals are either carried out using portable field equipment directly on the soil, or in the laboratory, after soil samples have been collected. The method is metal specific. There are different sensitivities for individual heavy metals, and therefore different detection limits. The equipment contains a radio-active source which decays over time. This means that the sensitivity of the instrument declines over time, and it should be maintained according to the manufacturer’s instructions.

Since the method is very sensitive to the soil type, field measurements should usually be checked with laboratory analyses.

There are various types of field methods where the type of contamination can be determined using solvents which produce a colour reaction which depends on the amount of contamination present. The different methods and the types of contamination which can be detected are described in more detail in the Guidelines on sampling and analysis of soil. /3/.

4.4.2 Laboratory analyses

As a rule, approved laboratory analyses should be carried out during contamination investigations. The analyses should identify potential contamination, quantify the substances, have a detection limit that corresponds to acceptance criteria (the detection limit should be no higher than 1/10 of the acceptance criteria) and have an acceptable accuracy (typically a standard deviation of 10-20 %).

Laboratory analyses can either be carried out as screening analyses, or as substance-specific analyses. Screening analyses are normally utilised when the contamination parameters are not known, and where it is important to investigate for a number of substances. When contamination is found, a number of samples can be examined using specific analyses. The different methods of analysis are listed in the table included in Appendix 4.7. For further details of laboratory analyses, see the Guidelines on sampling and analysis of soil /3/.

4.5 Collection of data regarding buildings

When investigating the indoor air, or before implementing remedial measures for volatile substances and landfill gases, it is necessary to examine existing buildings, cf. guidelines from the National Building and Housing Agency /7/.

Background information on buildings can be obtained from the local authority, from the Land Registry Office, from previous and current owners, from investigations, etc. Information should include:

- The age of the building
- The construction of the building
- Thickness of materials and floor construction in contact with the soil
- Presence of reinforcement and concrete quality
- Height of rooms
- Renovation works carried out
- Current and previous use of the building, production etc.
- Cables and pipes.

In a subsequent investigation, the extent of agreement between the background information obtained and the actual conditions is examined. Furthermore, the following is examined:

- Whether there are visible cracks in the floor /16/.
- The quality of the building construction immediately above the soil, e.g. whether pipe lead-ins are air tight /16/.

- Whether the building itself, its land use, fittings, activities in the building, or any storage can lead to contamination of the same type as that which may be under the building.
- The surroundings of the building for sources of air pollution which may cause the same type of contamination as that which may be under the building.
- Whether there are signs of dampness (or similar) from the ground.
- Smells and odours in the building
- Ventilation.

The above information is used in the overall assessment of the condition of the building, and in assessing the construction measures necessary to secure the quality of indoor air, see the Guideline on Surveying Air in Building Constructions /16/.

4.6 Geology, hydrogeology, and hydrology

During an investigation of contamination, geological and hydrogeological conditions should be elucidated. The relevant soil strata should be described geologically. In addition to utilising basic data, this description is prepared primarily on the basis of geological characterisation of soil samples from the investigation borings. These may be supplemented with geotechnical tests and geophysical measurements. The scope of the geological description is detailed in Appendix 4.8. For further information regarding the geological description, refer to DGF’s Guidelines on Sample Description /8/.

The hydrogeological description details relevant groundwater conditions. The groundwater aquifer is assessed on the basis of the geological description and groundwater observations in the wells. Furthermore, these can be supplemented with pump tests, cf. Section 4.2.5, and possibly groundwater models. Water flow in the individual groundwater aquifer can now be determined in the form of the potentiometric surface, flow direction, gradient, hydraulic parameters, and leakage.

Surface water recipients in the area should often be mapped during a contamination investigation. When the recipient is located close to the contaminated site, investigations should be made as to whether contamination has taken place. Contamination can occur through groundwater flow or surface run-off. In most investigations, contamination from groundwater is the most critical.

On the basis of the geological model and water-level measurements from the groundwater and from the recipient, assessments can be made as to whether the contamination can reach the recipient. If this is the case, the concentration contribution of the contaminants is calculated on the basis of analyses of groundwater samples collected close to the recipient, taking into consideration mechanisms such as mixing, degradation, and sorption.

5 Risk assessment

5.1 Definitions, procedures and data requirements

A risk assessment is an evaluation of environmental and health-related effects of contamination. The purpose is to assess the need for protective measures, since a specific risk assessment is a precondition for any such protective measures.

Risk assessments appraise specific circumstances and are based on information about actual contaminants, transport and exposure pathways as well as target groups at risk in each given situation. The risk assessment must be based on:

- The results of the investigations, including the nature and extent of contamination as well as prevailing geological, hydrogeological and hydrological conditions.
- A hazard assessment pertaining to the contaminants of interest.
- A survey of possible ways of transport and exposure pathways (vulnerability assessment).
- Knowledge of the target groups exposed.

A specific risk assessment will highlight interconnected ways and effects that may constitute a hazard to the target group. Further principles for carrying out a risk assessment may be found in Environmental Project 123/12/. In these guidelines, the main focus is on human exposure. On rare occasions, it may also prove necessary to take ecotoxicological aspects into consideration.

A hazard assessment is an review of the inherent characteristics of a potential contaminant. Qualitatively, a hazard may be described as carcinogenic, corrosive, toxic, etc. and effects may characterised as acute and more long term (chronic effects). Whenever possible, the hazard is quantified by determining the concentration level at which harmful effects arise.

Determining the inherent hazard of a given contamination incident entails a comprehensive assessment of toxicity, biodegradability, bioavailability, and mobility.

For a number of relevant substances, hazard assessments have already been carried out in connection with the setting of quality criteria and B-values. Data sheets and brief profiles of pertinent substances, their effects and the basis for each calculation have been prepared (9, 10, 11).

Vulnerability is assessed by considering the possible transport and exposure pathways as illustrated in Figure 5.1.

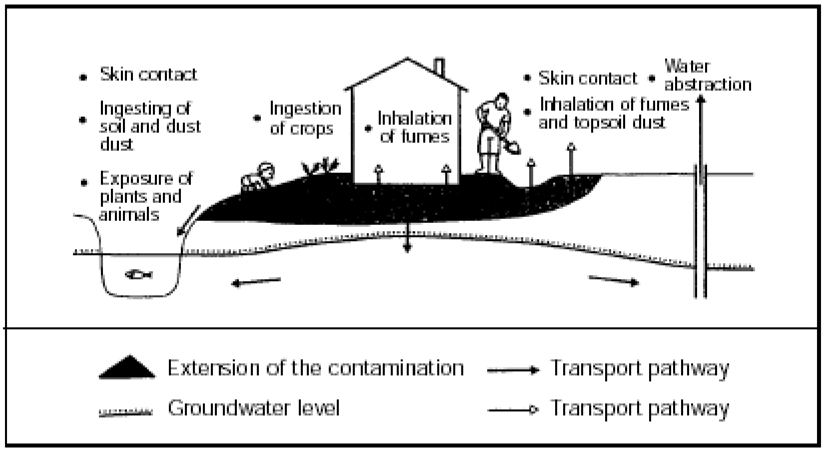


Figure 5.1
Transport and exposure pathways

Frequently, the vulnerability of the environment can be assessed by evaluating the three most important considerations; health considerations in connection with land use, groundwater protection considerations and considerations regarding surface-water recipients and soil.

The section dealing with investigations (section 4) mentions overall requirements regarding the parameters to be measured. It is therefore important that the risk assessment is planned as an integral part of the investigation work.

Soil contamination cannot be clearly distinguished from soil gas or groundwater contamination. In the saturated zone, the space between soil particles is filled with groundwater. The contaminants are in a state of dynamic equilibrium between soil particles and groundwater. Similarly, there is air and water between the soil particles in the unsaturated zone and the volatile substances will reach a dynamic equilibrium between soil, soil gas, and water.

Thus, while distinction between the contamination of soil, groundwater and soil gas may be difficult in a purely physical sense, it may still be useful to carry out separate risk assessments in connection with land use, groundwater and evaporation.

5.2 Land use

5.2.1 Definitions

When remediating a contaminated site, various soil types may be encountered: undisturbed soil that is contaminated, excavated soil that is either contaminated or non-contaminated, and soil conveyed from an external source. The situation is illustrated in figure 5.2.

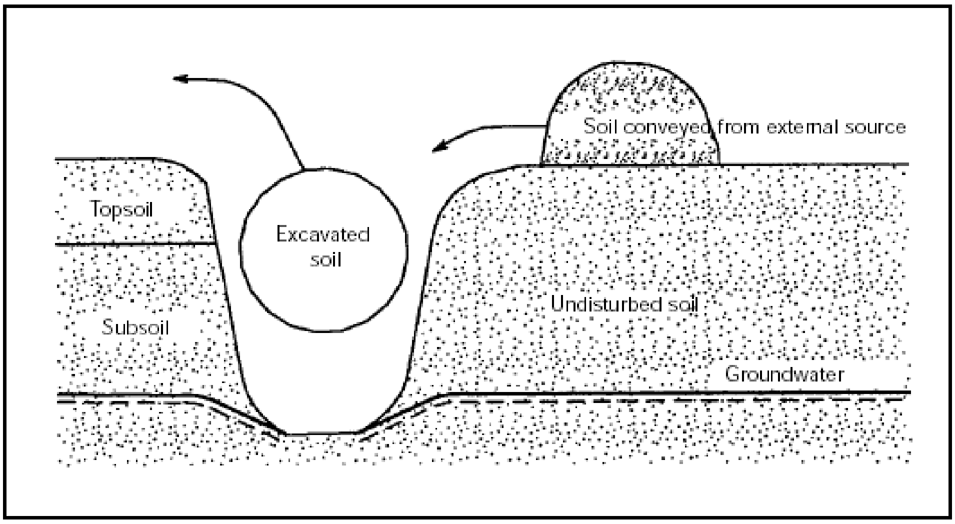


Figure 5.2
Outline of soil types which may be encountered in remediation of a contaminated site.

Much topsoil in urban areas is typically composed of fill.

These guidelines distinguish between:

- Soil which is used in excavations at the site, be it from other parts of the site, or soil conveyed from an external source or treated soil.
- Topsoil. This is the uppermost and thus the most sensitive soil with regard to surface-related activities. Topsoil thickness typically varies from 0.25 m to 1 m, depending on what the site is used for.
- Subsoil. This is the stratum between the underside of topsoil and the groundwater level.

When conducting risk assessment of contaminated sites, one must discriminate between the concepts of quality criteria, acceptance criteria and cut-off criteria. These concepts are defined in Figure 5.3.

In these guidelines, prime attention is given to human exposure. For ecotoxicological aspects, please refer to the Project on Soil and Groundwater from the Danish EPA, about ecotoxicological soil quality criteria /10/.

It should be noted that compliance with soil quality criteria does not automatically ensure compliance with criteria pertaining to groundwater or air. The quality criteria are described further in Chapter 6.

Quality criteria	The soil quality criteria are established for public health reasons. These standards are based on human toxicological assessments and knowledge as to which pathways of exposure are pertinent for substances in soil. In the case of certain substances, considerations of an aesthetic/hygienic nature, such as odour or appearance have also influenced the criteria. Anyone may make free and unlimited use of soil complying with the soil quality criteria, even in respect to highly sensitive purposes. Compliance with the soil quality criteria, however, does not automatically ensure compliance with groundwater or air criteria.
Acceptance criteria	These criteria mark the acceptable amount of each substance in the soil, given the land use pattern and physical location of the *specific* site in question. The acceptance criteria are based on a site specific risk assessment and depends on what the site is used for.
Cut-off criteria	For certain immobile and slowly degradable substances, not only soil quality criteria are set, but also a cut-off criterion for contamination of the upper layers of soil.
	For areas with highly sensitive land use, cut-off criteria denote thelevel of soil contamination above which complete cut-off of all contact with the soil is called for; entailing, say, a total clean-up or excavation.

Figure 5.3
Soil criteria definitions.

5.2.2 Exposure

Soil contamination may pose a threat to humans, animals or plants in cases of land use at ground-level. This problem will primarily involve topsoil.

For humans, the following exposure pathways exist:

- Ingestion soil.
- Eating crops grown on the soil.
- Skin contact with soil.
- Inhalation of soil particles.
- Inhalation of fumes from soil.

In addition to the effects on humans, exposure of plants and animals may take place..

Crucial exposure pathways will typically depend upon site use, since different land uses of a given site will lead to distinct patterns of activities. The following factors will be of significance in

- Accessibility of contamination
- Duration of exposure
- Exposure pathways
- Sensitivity of the user group

Sensitivity of various site uses is subdivided into three levels: highly sensitive, sensitive and non-sensitive (see Figure 5.4). The most sensitive land uses of contaminated sites involve an evident likelihood of children ingesting soil and/or the possibility of human exposure to unacceptable, health-impairing exposures through inhalation of fumes from volatile contaminants.

The depth of land use is the depth to which the soil is regularly used at a given site. Construction work deeper than the depth of land use may entail the need for appropriate measures.

Generally, the minimum depth of land use should be:

- 1 m. Where land use is highly sensitive, i.e. in the gardens of private homes and in day-care facilities, since soil may be frequently worked, etc., and edible crops may be grown.
- 0.5 m. In park areas and other publicly accessible spaces with varying, albeit primarily permanent planted areas.
- 0.25 m. Where an area is permanently consolidated or covered with grass, thus excluding any further earth works etc.

Circumstances may arise where the depth of land use exceeds the above or where, occasionally, the need to work at deeper layers arises. Such circumstances may occur when planting or digging up trees, or when carrying out excavation work in connection with construction activity.

The Environmental Project number 123 from the Danish EPA /12/ contains an overview of the patterns of exposure encountered in connection with many normal land uses This figure is reproduced, in a somewhat modified form, as Figure 5.4. Working from the above guidelines, a specific assessment is decided upon in each instance; i.e. an acceptance criterion is established.

In connection with soil remediation, specific clean-up depths may be proposed, based on the land use depth of the individual site. As a consequence, this can result in remediation at depths with are less than the depth of land use or depths that are greater than the depth of land use due to removal of vegetation, thawing, subsidence or evaporation to the surface from lower, contaminated soil strata (refer also to Section 5.2.3.).

Typically, this means that no general, differentiated acceptance criteria with respect to the duration of exposure for a specified land use can be determined for a planned remediation, Instead, the chosen parameter is the land use depth at which the soil quality criteria must be met.

Land use	Sensitivity	User group	Site characteristics	Daily duration of exposure	Ways of exposure		
					Inh.	SkC.	Ing.
Roads, etc	Non-sensitive	Healthy adults	Paved	Minutes	(+)	-	-
Industrial	Non-sensitive	Healthy adults	Buildings Car parks Grass	8 hours Minutes Minutes	++ (+) (+)	- - (+)	- - -
Office	Non-sensitive/ Sensitive	Healthy adults	Buildings Car parks Grass	8 hours Minutes Minutes	++ (+) (+)	- - (+)	- - -
Shops: foodstuffs other	Non-sensitive/ Sensitive	Healthy adults, children, pregnant women, elderly, the sick	Buildings	Employee/-customer 8 hours, 1 hour 8 hours, 1 hour	++ ++	- -	(+) -
Blocks of flats	Sensitive	Healthy adults, children, pregnant women, elderly, the sick	Buildings Car parks Grass Playgrounds	24 hours Minutes-hours 4-12 hours 4-12 hours	+++ (+) (+)	- - + +++	- - + +++
Private houses	Highly sensitive	Healthy adults, children, pregnant women, elderly, the sick	Buildings Gardens (grass) Flower beds Crops	24 hours 4-12 hours 4-12 hours 3/4 of the year	+++ (+) (+)	- + +	- + ++ ++
Allotment gardens	Sensitive	Healthy adults, children, pregnant women, elderly, the sick	Buildings Gardens (grass) Flower beds Crops	4-8 hours 1/4 of the year	+++ (+) (+)	- + +	- + ++ ++
Recreational areas	Sensitive	Healthy adults, children, pregnant women, elderly, the sick	Grass Playgrounds Flower beds Paths	3-5 hours 3-5 hours Minutes/hours Minutes	(+) + (+) (+)	+ +++ ++ (+)	+ +++ ++ -
Schools	Sensitive	Healthy adults, children of school age, pregnant women	Buildings Paved Grass	4-8 hours 2 hours 1 hour	++ - (+)	- - ++	- - (+)
Kindergartens	Highly sensitive	Healthy adults, children, pregnant women	Buildings Playgrounds Paved Car parks	8 hours 8 hours 8 hours Minutes	++ + - -	- +++ - -	- +++ - -
Nursing homes	Sensitive	Elderly, sick, healthy adults, pregnant women	Buildings Grass Fenced-in	24 hours 0-3 hours	+++ (+) -	- (+) -	- - -

- : no likelihood of exposure
(+) : slight likelihood of exposure
\+ : some likelihood of exposure
++ : greater likelihood of exposure
+++: great likelihood of exposure

Inh. Inhalation
SkC. Skin contact
Ing. Ingestion

Figure 5.4
Exposure patterns for various site uses

5.2.3 Acceptance criteria and remediation principles

An important element of risk assessment is to establish acceptance criteria for the contaminant with respect to the sensitivity of the specific land use.

For quite a number of substances, the Danish EPA has determined toxicological quality criteria for soil involving highly sensitive land use. These quality criteria are listed in Figure 6.1.

In the case of ten selected substances, soil contaminant cut-off criteria have been specified for highly sensitive land uses regarding the upper soil strata. The cut-off criteria indicate at which point compliance with basic advisory precautions will no longer reduce exposure to the same level that compliance with the soil quality criteria ensures.

The cut-off criteria are explained in Figure 6.2.

The soil quality criteria and cut-off criteria apply to the depth of land use. Compliance with the EPA's soil quality criteria does not preclude that near-surface contamination may pose a risk for recipients/groundwater or emit unacceptable evaporation to outdoor or indoor air (see Sections 5.3 and 5.4).

If remediation without subsequent regulation with a view to future land use is desired (equivalent to unconditional closure under the Waste Deposit Act), the soil quality criteria should be met down to roughly 3 m below ground level, unless the groundwater level is closer to the surface and therefore a limiting factor. There can, however, be some situations where it is necessary to remove or remediate deeper contamination due to the risk for contamination of groundwater or where there is a limited extra expense involved in removal of all of the contamination.

If land use and therefore the risk of exposure is regulated administratively, there will not be a need for physical remediation such as excavation, placement of geotextile, etc. The reason is that administrative regulation can ensure that no exposure to the contaminants occur.

I land use is regulated so restrictively that contact to contamination cannot and will not take place (working the soil such as construction, landscaping, etc.) or if the area is paved or has a permanent grass cover, then the depth of land use may be considered nil.

Possible deterioration or drying out of an area with permanent grass cover can lead to the formation of areas with bare soil and will require that entry to the area is forbidden until the grass cover is re-established. The administrative regulation will therefore demand continuing responsibilities and maintenance.

Reducing the depth of land use via administrative regulation, should take place after weighing the possible inconveniences with the cost of obtaining a greater depth of land use.

In practice, some areas will exist where a administrative regulation is not appropriate or where nil depth of land use cannot be ensured. For volatile substances, additional administrative regulation of an area forbidden entrance or residence may be required even in cases with a nil depth of land use. This is due true for instances where evaporation of the contaminant to the outdoor air presents a risk.

A risk assessment must identify *all* present and/or potential *land use conflicts.*

Based on the risk assessment, a decision is made as to the appropriate course of action, clean-up/replacement or advisory measures. Steps taken must eliminate *the present* conflict of land use.

In the practical performance of a remediation task, compliance with the acceptance criteria to the prescribed land use depth must be ensured.

It is acceptable to let residual contaminants remain at depths below the normal land use depth. In all cases, "Hot spots" containing high levels of contamination (e.g., gas purification residue, buried chemical depositions, drums and tanks formerly used for petrol or tar) should be removed or dealt with. There may be circumstances where it is not immediately practical or economically feasible to undertake a clean-up according to the acceptance criteria. Moderate conflicts of land use may be obviated by use of advisory precautions which ensure an acceptable reduction of exposure (see section 6.3).

It is also possible to replace the top 30 cm or use administrative regulation which reduces or prevents exposure with contaminants.

In cases of highly sensitive land use where conflicts of land use are resolved by replacing the top 30 cm with clean soil, defined as soil meeting soil quality criteria, the contaminated soil lying below must be separated from the clean soil using geotextile and/or an identification net.

The purpose of the geotextile is partly to prevent any contact with contaminated soil and partly to prevent any mixing of clean and contaminated soil. Information regarding the contamination deeper than the geotextile must be available to the public. Such information must be contained in an administrative system, such as the "Bygnings- og Boligregistret, BBR", and will thus be imparted to the owner in notifications of changes. Furthermore, any activities deeper than the geotextile will require compliance with special precautions.

Whenever investigations show that soil quality criteria at depths exceeding 30 cm are exceeded while criteria from ground level to 30 cm have been met, highly sensitive land use will require the removal of the upper 30 cm and the use of geotextile before the land is reinstated for use.

I areas with sensitive land use, including parks and park-like areas, it is not always necessary to remove contaminated soil if administrative regulation can limit exposure to an acceptable level or prevent exposure altogether.

Familiarity with background levels of naturally occurring substances as well as xenobiotic substances are important elements in any health assessment.

Thus, soil quality criteria are not to be uncritically employed in instances where the natural background level of a given substance exceeds the soil quality criterion. This may be the case for certain metals.

5.2.4 Assessment of soil contamination

In the following, principles regarding the assessment of soil contamination in the uppermost metre as it relates to land use conflict are described for the case of highly sensitive land use.

The starting point is that any small, yet consistent, transgression of the soil quality criteria in an area necessitates remediation (advisory precautions, cut-off, excavation, or the like).

If the soil quality criteria is exceeded at a potentially contaminated site, the contamination is mapped and remediation takes place.

If there is a tendency towards falling concentrations as a function of distance from the highest concentrations, a hot spot likely exists, even if historical records do not indicate a potential contamination source at the location. The area must be mapped, and the contamination remediated.

A single value which exceeds the criteria does not constitute grounds for declaring a hot spot, and can therefore not prompt a mapping investigation. A hot spot must be verified by several analyses.

In some cases, it may be possible to identify a characteristic fill layer with concentrations that exceed soil quality criteria. Fill layers may be visually recognised in soil profiles by the presence of slag, certain types of construction waste, soil type, colour and so forth. The extent of the fill layer not in compliance with soil quality criteria must be mapped, and remediation initiated.

If transgressions of criteria occur randomly over the entire area - thus precluding the existence of a hot spot or specific contaminated layer - the above procedure cannot be followed.

In the event of such diffuse contamination, assessment is carried out according to two different procedures, depending on whether the contamination involves a substance where a chronic or sub-chronic effect has been the determining factor in setting soil quality criteria, or whether it involves a substance where an acute effect has prompted the quality criteria set.

Substances in which a chronic, harmful effect is the determining factor in setting the quality criteria (e.g. lead, cadmium, benz(a)pyrene and total PAH), the area may be used for highly sensitive land use only if the average of all tests conducted lies below the soil quality criteria that have been set.

Substances in which an acute harmful effect is decisive in setting the quality criteria (like arsenic, or nickel), the area may be used for highly sensitive land use, provided both of the following criteria are met:

- The average of all samples lies below the quality criteria set.
- A maximum of 10 per cent of all samples show results exceeding the established soil quality criteria; while none of these exceed soil quality criteria by more than 50 per cent.

Table 6.1 identifies whether an acute or a chronic effect has prompted the establishment of the quality criteria for the individual substances.

The above principles assume that a contaminated area that is relevant to the land use is investigated. In other words, samples from areas known to be uncontaminated may not be included.

As an absolute minimum, analyses comparable to level 1 in the Guidelines on sampling and analysis of soil should be performed /3/.

5.3 Evaporation, including landfill gas

In buildings or outdoor areas above contaminated ground, there is a risk of an unacceptable influence on the indoor or outdoor air from contaminants in the soil or groundwater. This may be caused by volatile substances. Among the substances present at contaminated sites, the risks are greatest when contamination involves highly volatile organic solvents, including chlorinated solvents. Furthermore, methane rising from landfills may constitute a risk of explosion.

Below, different methods of risk assessment with respect to outdoor air, indoor air and landfill gasses, are dealt with separately.

All methods involve different procedural steps. Several of these steps implement different theoretical methods of computation for the interrelationship between the soil contaminant content and the evaporation which may be expected from this source and its conveyance to the outdoor or indoor air. The models are described in Appendix 5.2-5.4. These models provide a conservative estimate of the contamination contributed to the outdoor or indoor air. The models have been included in these guidelines because they are fairly simple; they may only be used under the circumstances and provisos outlined in the Appendices.

More advanced models which allow for degradation and water infiltration have been described in a project concerning the dynamics of chemical substances in soil and groundwater /13/ and will not be dealt with further here.

In addition to factors like the nature and concentration of the contaminants in question, many other factors determine how contaminants evaporate into the air, be it inside or outside a building. These factors include:

- Depth of contamination
- Porosity and water content of soil layers
- Building design and the materials used for its construction
- Temperature and pressure gradients surrounding the building
- Building ventilation

Interpretation of measurements and calculation results may be made difficult by the presence of other types of indoor or outdoor contamination, e.g., degassing from building materials, furniture, carpeting etc, smoking, traffic, nearby industrial plants, etc.

5.3.1 Outdoor air

Volatile contamination poses a risk of unacceptable evaporation to free areas (vacant or unpaved areas).

Figure 5.5 displays a flowchart delineating a model for handling risk assessment with regard to the outdoor air at contaminated sites designated for a highly sensitive land use.

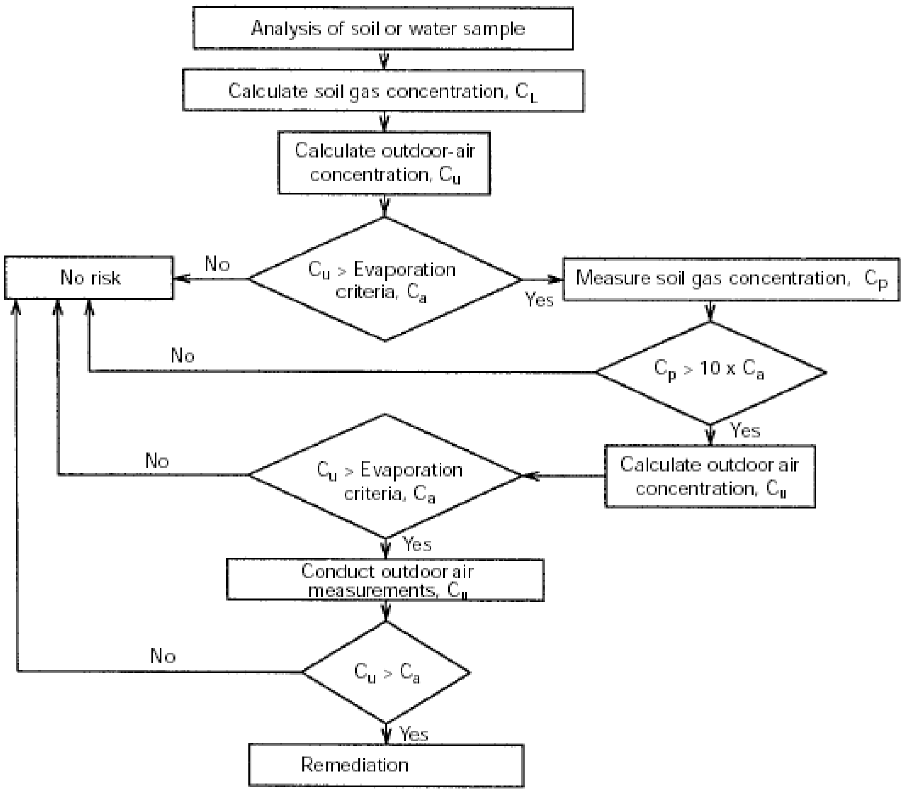


Figure 5.5
Flowchart for outdoor-air risk assessment

This model is based on analysis results obtained through soil and/or water samples and involves four different steps:

1) Calculation of soil gas and outdoor air concentrations
2) Measurements of soil gas concentration
3) Calculation of outdoor air concentration
4) Conducting measurements of the outdoor air

As is evident from Figure 5.5, each step in the assessment will either lead to the contamination being deemed non-problematic; or it will lead to the next step in the flowchart. Below are guidelines for each step:

The soil gas concentration, C_p, is determined under the assumption that the phase distribution (i.e. the distribution of individual contaminants between soil gas, pore water, a phase absorbed in the soil matrix, and, occasionally, a separate contamination phase) may be calculated using the fugacity principle.

Formulae for calculating contamination distribution in the soil may be found in Section 3.1 of Appendix 5.3.

Parameters included in the calculation of phase distribution are:

- Soil type, soil porosity, water content, grain density, total density, calculated temperature and organic content.
- Molecular weight, soil-to-water ratio, vapour pressure and solubility

An example showing the calculation of evaporation of volatile substances from soil can be found in Appendix 5.4.

Chemical data for selected substances are given in Appendix 5.5. Standard data pertaining to soil types is listed in Appendix 5.3 (table 1, section 4).

Risk assessment is based on the principle that evaporation from soil to the air above must not exceed the evaporation criteria, C_a. Evaporation criteria for several substances may be found in Section 6.6; this section also demonstrates how evaporation criteria for substances other than those listed may be obtained.

Based on Danish EPA experience, the above criteria are considered to be met, provided soil gas concentrations remain less than 10 times the evaporation criteria.

For volatile substances, calculations of the soil gas concentrations in actual soil contamination cases tend to show that soil gas concentration exceed the evaporation criteria by at least 10 times.

Consequently, it is necessary to calculate how much the contamination contributes to the outdoor air.

Formulae for calculating the diffusive contribution to the outdoor air may be found in Section 3.3 of Appendix 5.3. Parameters included in the calculation are:

- Total porosity, water content, soil type, and thickness of each diffusion-inhibiting soil layer.
- Diffusion coefficient for the substance.
- Vertical mixing height.
- Wind velocity.

As a minimum requirement, the type of soil and the thickness of the diffusion-inhibiting layers of soil must be determined. Total porosity and water content may be estimated, based on the type of soil in question. Standard data may be used for vertical mixing height and wind velocity.

If the calculated, outdoor-air contribution exceeds the evaporation criteria, C_a, soil gas or outdoor-air measurements will have to be carried out.

In this model, estimates of soil gas concentration are very conservatively. Therefore, if the calculated outdoor air concentration exceeds the evaporation criteria, the soil gas concentration should be measured.

Soil gas measurements of organic gasses must be carried out according to guidelines laid down in Section 4.3.1 and Appendix 4.4.

If measurements show that the soil gas concentration, $C_{p,}$ exceeds the evaporation criteria by more than ten times, it cannot be ruled out that the contamination is problematic as regards the outdoor climate. Calculations that determine outdoor-air effects are then carried out as described above.

Should measurements indicate that outdoor air concentrations, C_u, exceed the evaporation criteria, it cannot be ruled out that the underlying contamination constitutes a problem and demands remediation. As mentioned previously, heavily-trafficked urban areas or industrial zones may have very high background levels.

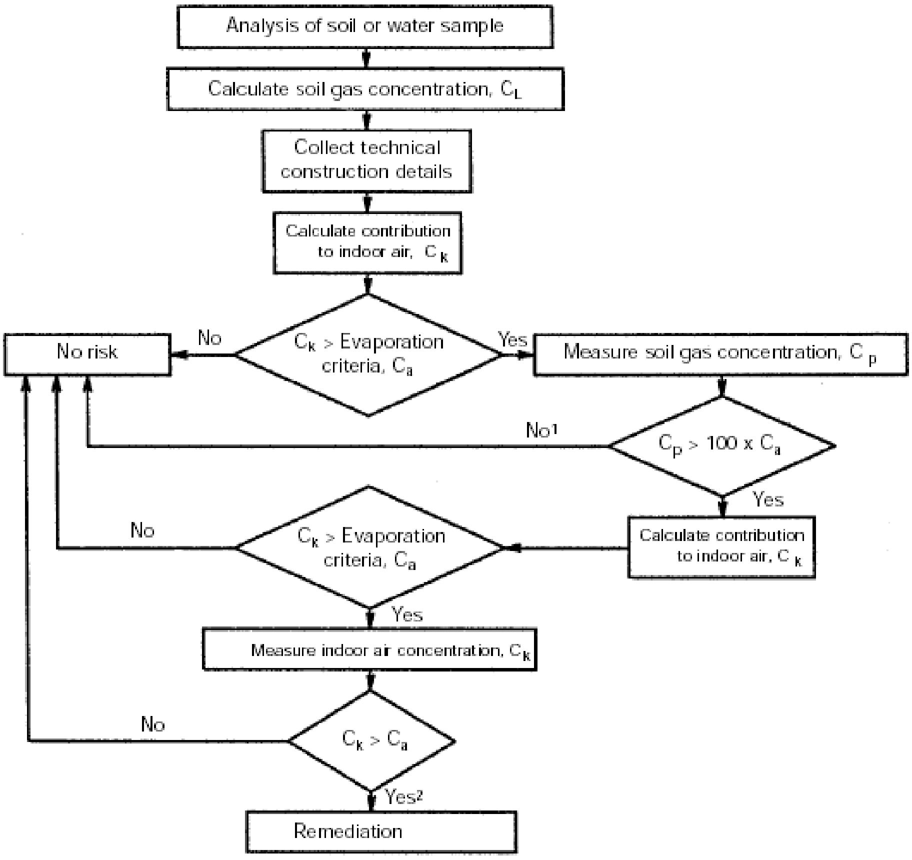
Before the final decision to remediate a specific site is made, it must be determined if it is probable that the increased outdoor air concentration is in fact due to soil or groundwater contamination. This may be done by comparing concentrations with outdoor air measurements carried out at (uncontaminated) neighbouring sites.

5.3.2 Indoor air

As with outdoor air (Section 5.3.1), risk assessment of indoor air in buildings located on plots allocated for residential, institutional or office purposes is based on the proviso that the evaporation emanating from the underlying contaminants into the indoor air must not exceed the evaporation criteria, C_a (see Section 6.6). The evaporation criteria C_a are not indoor air limits, but limit values applying solely to the maximum contribution to indoor air permissible from underlying contaminants.

In the case of non-sensitive land use, e.g. production facilities, other criteria for permissible evaporation levels are established for substances involved in production processes.

Figure 5.6 displays in a flowchart how to manage risk assessment of indoor air.



1: Assuming the concrete floor is without visible cracks
2: Assuming the contamination stems form contaminated soil or groundwater

Figure 5.6
Flowchart for risk assessment of indoor air

This model is based on the results of analyses of soil and/or water samples and operates with four different steps:

1) Calculating the soil gas concentration and evaporation into the indoor air
2) Measuring the soil gas concentration
3) Calculating the concentration contribution to indoor air
4) Conducting measurements of indoor air

As can be seen from Figure 5.6, each step in the assessment will either lead to the contamination being deemed non-problematic; or lead to the next step in the model.

Below are guidelines for each step in the flowchart:

Calculation of the soil gas concentration, C_L, is identical to the first step of the model for outdoor risk assessment. For details, please refer to section 5.3.1.

The majority of Danish residential and institutional buildings are constructed using flooring that contains a layer of concrete and passive ventilation of approximately 0.3 times an hour for residential buildings and approximately 2.0 times an hour for institutions.

Therefore, a dilution factor of 100 is considered a conservative estimate of the reduction in the contribution of soil gas concentration to the indoor air concentration.

If the calculated soil gas concentration, C_L, is less than 100 times the evaporation criteria, the underlying contamination is thus considered non-problematic. The conservative reduction factor of 100 is based on the building practices outlined above. Therefore, this reduction factor cannot be applied to buildings with wooden flooring, or concrete floors with large, visible cracks.

As mentioned previously, the calculation of soil gas concentration chosen in this model is very much on the safe side. When applied, a calculation of volatile substances will tend to be at least 100 times larger than the evaporation criteria, meaning that the contribution to the indoor air will have to be calculated.

Since the calculation of the soil gas concentration has been set on the safe side, the most expedient procedure will often be to start by measuring the concentration in the soil gas (refer to the guidelines in Section 4.3.1 and Appendix 4.4). The indoor air contribution may be calculated or the indoor air concentration may be measured.

If the measured soil gas concentration under the floor, $C_{p',}$ is higher than 100 times the evaporation criteria, it cannot be ruled out that the contamination is in excess of the evaporation criteria.

If the concentration in soil gas is calculated, the contribution to the indoor air must be calculated; likewise, this contribution is calculated if measurements of soil gas concentration prove to exceed evaporation criteria by a factor of 100 or more.

In order to undertake a practical calculation of the concentration contributing to the indoor air, technical construction details must be collected.

Concrete can be divided into four environmental classes:

- aggressive environmental class
- moderate environmental class
- passive environmental class
- non-reinforced concrete with no environmental class

The diffusion contribution through concrete is primarily dependent on the environmental class of the concrete and the aggregates porosity /54/. The contribution to convection is primarily dependent on the reinforcement.

In these calculations, a distinction is made between reinforced concrete flooring laid in accordance with the Radon guidelines /14/ and non-reinforced concrete flooring.

Technical construction details may be procured from original construction plans and descriptions as well as from on-site measurements. Methods for conducting air exchange measurements as well as crack and fissure measurements may be found in publications from the National Housing and Building Agency (*Bygge- og Boligstyrelsen)* /7, 16/.

Where reinforced concrete flooring is concerned, numerous parameters are included in the following calculations. Since collecting all this information will probably not be possible, and since it is not all of equal importance, the relative “importance” of the various parameters is listed in Figure 5.7.

Great importance	Thickness of the concrete Age of the concrete Concrete air humidity when poured and hardened
Lesser importance	Pressure variances across concrete cover Height to the building’s ceiling Air exchange of the building
Slight importance	Reinforcement used in concrete Cement content of concrete The water/cement ratio of the concrete

Figure 5.7
Overview of input parameters for reinforced concrete flooring

The minimum requirement is for concrete thickness to be measured. For concrete age, 20 years is used in the risk assessment; however, the actual age may be used if measured and calculated concentrations are to be compared. Concrete air humidity at the time of pouring and hardening will rarely have been ascertained in the case of existing concrete coverings, but may be managed for new concrete coverings.

In the case of non-reinforced concrete coverings, information may be collected on the thickness of the concrete, ceiling height, air exchange, floor area as well as the lengths and widths of cracks.

When calculating a contaminant’s contribution to the indoor air, one must calibrate the diffusive as well as the convective contribution. Formulae for calculating the diffusive concentration contribution to the indoor air are found in Section 3.4 of Appendix 5.3.

Parameters for these calculations include:

- Material constant and thickness for each of the diffusion-inhibiting layers of flooring.
- Ceiling height and air exchange for each ventilated room, as well as porosity in the case of ventilated drainage.
- Year of construction of the building/age of the flooring.

A set of formulae for calculating the convective concentration contribution to the indoor air through reinforced concrete flooring can be found in Section 3.5 of Appendix 5.3.

For an overview of calculation parameters, please refer to Figure 5.7. Standard data for concrete coverings and buildings may be found in Tables 2 and 3 of Appendix 5.1, section 4.

Appendix 5.4 provides an example of calculation of the diffusive and convective contributions.

If the calculated contribution to the indoor air, C_i, (the sum of the diffusive and the convective contributions) exceeds the evaporation criteria, indoor air measurements may be undertaken, or the basis of calculation may be enhanced by measuring the soil gas concentration.

Measurements of the indoor air should be carried out in accordance with the guidelines issued by the Housing and Building Agency /7/.

A considerable number of volatile organic substances may be expected to contribute to the background level in Danish buildings. A list of background levels can be found in G.M. Nielsen et al. /18/ and addition information is found in a report from the Housing and Building Agency /15/.

Thus, on the sole basis of measuring the indoor air concentration, C_i', it remains difficult to assess whether the concentration contribution from the residual contaminants exceed to the evaporation criteria.

Before a final decision is made to initiate remediation, it must therefore be determined whether it is likely that indoor concentrations exceed limits due to soil or groundwater contamination.

A comparison of the measured indoor air concentration must be made with the median value of the background levels, combined with a thorough investigation of plausible "additional" contributions to indoor levels stemming from building materials, furniture and fixtures, leisure activities or smoking.

5.3.3 Landfill gas

Sites formerly used for landfills may have problems with:

- Rising methane gas which may pose an explosion hazard for buildings and "hollow" installations at the landfill or in its immediate vicinity.
- Contaminants (typically heavy metals) in topsoil layers, due to inadequate covering of the former landfill.
- Leaching of percolate containing many different contaminants which may contaminate groundwater and/or surface water.

While the last two situations can be assessed as described in Sections 5.2 and 5.4, the assessment of explosion risk is dealt with separately in this section.

On sites where biodegradable waste has been landfilled, there is a potential risk of the production of landfill gas. Types of waste belonging to this category include household waste, garden waste, commercial and industrial waste, slaughterhouse waste, and lumber waste.

Disposal of soil, concrete, demolition rubble, chunks of asphalt and similar waste do not give rise to gas production.

An important step in the risk assessment is to identify the area at the landfill site where gas is produced. A figure outlining this principle is shown in Figure 5.8.

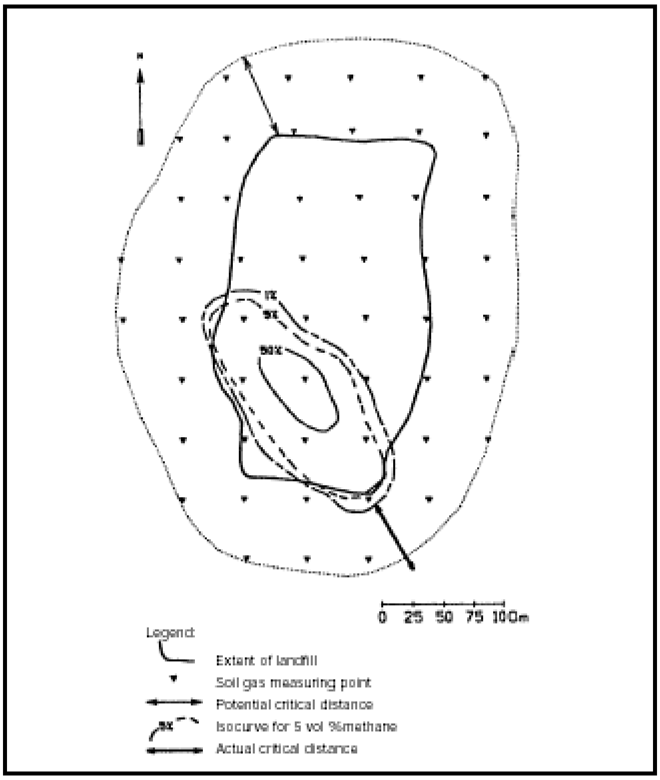


Figure 5.8
Illustration showing a method for identifying the gas-producing area at a landfill.

Figure 5.8 illustrates an assessment based on the results of soil gas measurements for landfill gas. For a description of the survey method, please refer to Figure 4.1 with its explanatory notes in the subsection “Landfill gas” in Section 4.3.2.

Based on measurement results, Figure 5.8 displays isocurves for concentrations of 50, 5 and 1 per cent methane gas (by vol.). The gas-producing area is defined as the demarcated section where methane concentration in soil gas exceeds 1 per cent (by vol.) The actual risk distance for critical gas emission must be measured from the 1 per cent (by vol.) limit.

Gas production at a former landfill site will subside over time. Besides the amount of time elapsed since the specific disposal, source terms will depend on a number of factors. The most important factors are the composition of waste and the amount of waste. Appendix 5.1 provides an empirical calculation model for attaining estimations of the gas production rate and potential residual gas emissions (source term).

Assessments of the risk of explosion due to gas penetrating into buildings within the critical area are based on the source term and technical construction information regarding the building.

A distinction is made between buildings situated within and outside the gas-producing area.
In the case of buildings situated on top of the gas-producing area, the diffusive and convective contributions to indoor air may becalculated using the same sets of formulae described under Section 5.3.2.

Here, one must decide what “worst-case over pressure” is envisioned in the soil gas below the building. Intervals for measured values of over pressure at landfills are described in Table 1 of Appendix 5.2. In this table there are also empirical figures relating to air permeability and air porosity for various types of soil. Furthermore, there are values for the dynamic viscosity of air, methane and carbon dioxide.

In the case of buildings situated outside the gas-producing area, the risk assessment may be based on a worst-case scenario, which would entail containment of the surface, due to a prolonged period of freezing weather or precipitation for example.

For a building located at a given distance from the gas-producing area, the attenuation factor, i.e. the relationship between the indoor air concentration in the building and the soil gas concentration at the landfill in a state of equilibrium (dilution factors) may be calculated using the convection model described in Appendix 5.2.

Furthermore, calculations are made to establish the time needed to attain the pressure gradient and equilibrium concentration.

The indoor air concentration of methane, once established, should not exceed 1 per cent (by vol.), this being the typical alarm level for gas monitoring equipment. Besides providing an assessment as to whether the alarm level might theoretically be exceeded, the assessment must also address whether the worst-case scenario does in fact represent a realistic risk. This latter question may be assessed on the basis of the residual gas potential, cf. Appendix 5.1., and the time required for establishing equilibrium (equation 3 in Appendix 5.2).

5.4 Groundwater

5.4.1 General remarks on groundwater risk assessment

The purpose of a risk assessment is to evaluate whether a given instance of contamination of soil or of secondary groundwater contributes contamination to the primary groundwater to such a degree that the groundwater criterion for the contaminant in question is exceeded.

Performing a risk assessment and subsequently remediating if necessary, shall ensure that groundwater resources remain pure; i.e. that groundwater criteria are met. Thus, the ultimate goal is that water supply wells may be placed without restrictions and abstract pure groundwater. This will also ensure that the flow of groundwater to recipients remains pure.

Groundwater criteria have been set at levels ensuring that groundwater may be utilised as potable water after ordinary, traditional water treatment.

Before a risk assessment can be undertaken, it is necessary to collect various field data from soil and groundwater. These data are collated in the survey phase which forms the basis of the actual risk assessment. Several of the parameters that are to be determined may vary, even within very small distances, while others are more difficult to determine. Thus, use of regional data or text book data may be required.

Any risk assessment should be based on the precautionary principle. In other words, one should select conservative data and values - data which are on the safe side. In effect, this means that one should calculate more conservatively when the parameters used are estimates, when parameters are regional rather than local, or when, for other reasons, parameters are disputable. The more data available for the site in question, the less conservative the estimates need be. The precautionary principle entails that the result of a risk assessment will amount to an overestimation of the contamination risk, erring to the side which benefits the groundwater.

In conclusion, the characteristics of the soil and secondary groundwater contamination must be determined in order to determine the contribution to the primary groundwater.

Risk assessment is based on the principle that the groundwater zone containing the highest concentration must comply with groundwater criteria.

If the contaminants are also naturally occurring substances, as is the case with many metals, the natural background level must be included in the risk assessment. This means that the contamination contribution which stems from the soil contamination must be even less in order to ensure compliance with groundwater criteria.

If a second site is the source of xenobiotic contamination stemming from human activity, this concentration should not be included in the risk assessment. Contamination with xenobiotic substance from other sources than the site currently undergoing risk assessment is thus not to be considered.

A risk assessment must not be more lenient simply because the aquifer is already contaminated. Even where an aquifer is already heavily contaminated, and thus unsuitable for groundwater abstraction, it is unacceptable that contamination deriving from a new contamination incident does not adhere to groundwater quality criteria.

The objective of a risk assessment is to assess whether a given instance of soil contamination (or of secondary groundwater contamination) contaminates groundwater resources, or may be expected to contaminate such resources in the short or long term.

In this context, it should be noted that the groundwater quality criteria are independent of soil quality criteria, since adherence to soil quality criteria will not automatically ensure adherence to groundwater criteria (and vice versa); see also Section 6.6.

The background for carrying out a risk assessment will typically be investigations that have established a case of soil contamination, but have not established any contamination of primary groundwater.

The reason that no contamination of primary groundwater is ascertained may be that the investigations did not extend to the groundwater, thus making it undetermined whether it is in fact contaminated. In other cases, the reason may be that there have been no drillings to the groundwater in the most highly contaminated area; perhaps because of the inherent danger of further contaminating the groundwater during that process.

Also, wells drilled downgradient of the contamination may show non-contaminated groundwater or very low levels of contamination. On this basis, it cannot be ruled out that the contamination in question poses a risk. There is the possibility that the maximum source term has not yet been reached, or the well may not be situated in the contamination plume, since it is very difficult in practice to locate a contamination plume downgradient.

The result of a risk assessment of soil contamination may be supplementary investigations which can forming the basis for a subsequent, more extensive risk assessment before deciding whether remediation is needed.

Instances of contamination in secondary aquifers may be dealt with in a similar manner.

5.4.1.1 Definitions

Leaching of contaminants from the contamination source, measured as flux (for example in kg of substance/year) is defined as the source term. The source term varies over time and place.

The source term concentration is the maximum concentration of contaminants (over time and place) released to the pore water from the contamination source.

The source term concentration is determined as follows:

- Measuring the concentration in pore water in the unsaturated zone immediately under the contaminated soil; e.g. in a conduit or tank pit.
- Utilising the contaminant's maximum solubility.
- This requires, that the mass of contamination is of a certain size. In any case, the use of maximum solubility generally gives a gross over estimation of the source term concentration.
- In many situations, only the concentration of contaminants in the soil (or possibly in the soil gas) will be known. In these cases, the source term concentration is calculated based on an assumption regarding equilibrium between phase distribution in soil, water, and air (the fugacity principle), as described in Appendix 5.3.

An example of such a calculation of the source term concentration can be found in Appendix 5.9.

A sudden release refers to a contamination pulse with a short duration, which is then brought to an end. In the unsaturated zone, an example of a sudden release might be a barrel that ruptures.

A continuous release refers to a contamination of large mass (in theory, of infinite mass) which therefore produces an continual stream of contamination. Examples of this could be a landfill, a major instance of soil contamination, or a leaking tank that is regularly refilled.

It should be noted that sudden releases in the unsaturated zone must often be considered a source of continuous release to the saturated groundwater zone. An example: a ruptured oil tank may result in a sudden release to the unsaturated zone; but the depth to the groundwater level will often require considerable time before the contamination front leaches to the saturated zone. As oil contaminants in leaching pore water also dissolve slowly, oil from such a ruptured tank will typically leach to the saturated groundwater zone over decades. Consequently, leaching of oil contaminants from a ruptured oil tank to the saturated groundwater zone must be regarded as a continuous release.

Primary aquifers are larger, interconnected aquifers of regional significance from which groundwater is or can be abstracted. In major parts of Zealand, the primary aquifers are limestone aquifers, which are often confined, while in Jutland, a considerable number of aquifers are in unconfined meltwater sand.

Secondary aquifers are secondary from the point of view of groundwater abstraction. Secondary aquifers are more shallow than primary aquifers, and are often unconfined.

In the context of the spreading of contamination and risk assessment, a secondary aquifer is given equal priority to a primary aquifer if significant spreading of contamination from the secondary to the primary aquifers and/or recipients is possible; or if the secondary aquifer is or can be used for water supply purposes.

If the concentration of a contaminant exceeds its maximum solubility, *NAPL* a separate phase known as a non-aqueous phase liquid (NAPL) will be present.

If contaminant concentrations which indicate presence of NAPL are found, there will, as a starting point, always be a risk. As a minimum, the NAPL should be removed. Risk assessment thus evaluate only dissolved contaminants.

When contaminants reach very large concentrations, the large differences in the density between the contamination plume and the surrounding, pure groundwater that the flow is directed by these density differences. The term used is for this is density flow. Density flow has been observed, for example, in the flow of NAPLs and the flow of landfill leachate.

The method that follows for stepwise risk assessment is not applicable to contaminants which spread by density flow.

A contamination risk is deemed to exist for groundwater if the final calculated resultant concentrations from the stepwise risk assessment exceed the groundwater criteria for the contaminant in question.

The contribution of contamination to a primary aquifer which results from a given soil contamination or from contamination in a secondary aquifer (and thus the calculated resultant concentration in primary aquifer), will depend on a number of factors which vary from site to site. The most significant factors are as follows:

- The contamination scenario, i.e. type of substance (mobility/degradability and other substance characteristics) as well as the contaminant concentration and area.
- Geology and hydrogeology, i.e. types of sediment (clay/sand/lime, organic content, hydraulic conductivity, effective porosity), net

precipitation/groundwater recharge, groundwater gradient, pressure gradient between the secondary and the primary aquifer, as well as redox conditions.

The following processes take place in the unsaturated as well as in the saturated zone and will lead to a reduction of the contaminant concentration in groundwater in the primary aquifer:

- Sorption: This effect is of particular relevance for sudden releases. In the case of larger-scale, continuous contamination release, the soil sorption capacity will gradually deplete around the waste site.

In the risk assessment that is described in the following, sorption is taken into consideration only in the saturated zone (risk assessment step 3).

- Dispersion: As a direct process, this effect is of greatest significance for the saturated zone, since the flow velocity in the unsaturated zone is relatively low, approximately 0,25-2 m/year, while typical pore water flow velocity is on the order of 10-1000 m/year in the saturated zone.

In the risk assessment that is described in the following, dispersion is taken into consideration only in the saturated zone.

As an indirect process, dispersion has significance for the transport time through the unsaturated zone. Because a major mechanism for contaminant removal can be biological degradation, where time is a crucial factor, dispersion will indirectly have a crucial influence on the removal of contamination.

- Natural degradation: is significant in the unsaturated as well as in the saturated zones, but is included in the risk assessment only in the saturated zone.

Appendix 5.8 lists first-order degradation constants, which are considered to represent degradation under natural condition in Denmark.

In practice, locating a groundwater contamination plume is often difficult (even just tens of metres downgradient of the contamination source), which makes monitoring of the expected degradation very difficult in the saturated zone.

First-order degradation constants for the unsaturated zone are not yet publicised and well-documented methods for monitoring possible degradation are lacking.

5.4.2 Stepwise risk assessment method

Outlined below is a stepwise risk assessment of soil contamination, carried out with respect to an aquifer. The stepwise method of conducting a risk assessment has been designed to achieve a balance between the amount of data required from the contaminated site, and the complexity of the model. With small amounts of data, a simple model is applied, and with larger amounts of data, a more advanced model can be utilised.

A Step 1 risk analyses will produce a more conservative result than the other steps, even if the same basic data is used.

The stepwise risk analysis is illustrated in Figure 5.9. Steps 1 and 2 are conservative and simple methods which can be utilised even if only limited site-specific data from the contaminated site are available. Step 1 is a near-source mixing model, where calculations are based on the mixing of contaminant leaching from the unsaturated zone into the upper 0.25 m of the water table. Steps 2 and 3 assess the resultant contaminant concentration at greater distances from the contaminant source. Step 2 is a mixing model where the thickness of the mixing layer (d_m) is calculated on the basis of dispersivity, pore water velocity, and mixing time. Step 3 is a mixing model like Step 2, but calculations take into account a reduction in the concentration of contaminants as a consequence of sorption, dispersion, and degradation in the saturated zone.

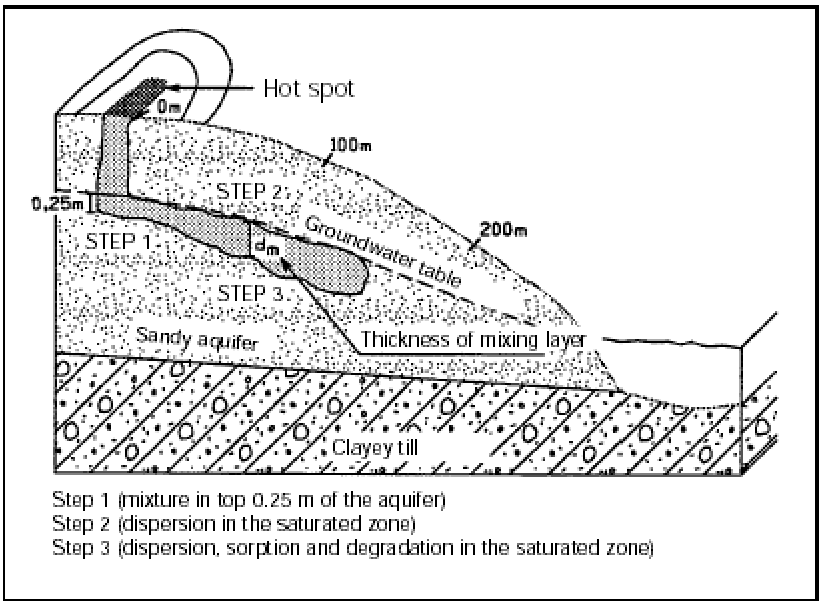


Figure 5.9
Illustration of risk analysis steps 1-3

In Step 3 of the risk assessment, sorption and degradation may be included in the calculations. It is not possible, however, to identify a degradation constant or a sorption coefficients for substance mixtures, such as engine oil or petrol. In these cases, it is necessary to make calculations without sorption and degradation for the substance mixture. Sorption and degradation may then be included in the calculations for a specific environmentally harmful substance, i.e. a substance with the slowest degradation rate, lowest groundwater quality criteria, and lowest sorption constant in relation to the actual concentration. Since the concentration of specific substances in a mixture is a significant factor for the selection of the most environmentally harmful substance, general guidelines can not be formulated.

Examples of specific use of the stepwise risk assessment can be found in Appendix 5.7.

5.4.2.1 Step 1. Near-source mixing model:

In near-source risk assessment, the resultant contaminant concentration in groundwater is calculated immediately under the area of contamination.

The conservative calculation which is used, assumes that the pore water at the bottom of the unsaturated zone has a concentration of contaminants which is equal to the concentration at source. After this, the calculation assumes that the contamination is mixed in the upper 0.25 m of the aquifer (if the aquifer thickness is less than 0.25 m's, the actual thickness is used).

Step 1 of the risk assessment does not account for sorption, dispersion and degradation. It is assumed that the aquifer is homogenous and isotropic.

Calculation of the content of contaminants in groundwater is performed as a mass balance for the upper part of the aquifer. The concentration of contaminants C_1 in the upper part of the aquifer is (cf. Appendix 5.6):

$$C_1 = \frac{A \cdot N \cdot C_0 + B \cdot 0.25[\mathrm{m}] \cdot k \cdot i \cdot Cg}{A \cdot N + B \cdot 0.25[\mathrm{m}] \cdot k \cdot i}$$

where:

C_0 is the concentration at the source $[ML^{-3}]$,
C_g is the natural background content of contaminants in the groundwater $[ML^{-3}]$,
A is the size of the contaminated area $[L^2]$,
N is the net infiltration $[LT^{-1}]$,
B is the breadth of the contaminated area (calculated in relation to the direction of groundwater flow) [L],
k is the aquifer's hydraulic conductivity $[LT^{-1}]$, and
i is the hydraulic gradient [unitless].

Specific examples of performing risk assessments can be found in Appendix 5.7. Examples of standard data, including typical values for hydraulic conductivity k, can be found in Appendix 5.8.

The resulting contaminant concentration in the upper 0.25 m of the saturated zone can also be determined directly through analysis of groundwater collected from a screen (with a length of 0.25 m) placed at the top of the aquifer. In connection with the risk assessment, the highest of the measured concentrations should be used.

When using measured concentrations, it is important to make a judgement as to whether the case involves a sudden release to the groundwater (which has been discontinued) or a continuous release (or a release which takes place over an extended time period) which gives a contribution to contamination of the groundwater aquifer. If groundwater is to be measured, it is a prerequisite that the maximum concentration has already reached the groundwater and that the front of contamination is not on its way through the unsaturated zone, resulting in higher concentrations at a later date.

The local geology must be known well enough to place the monitoring well in an optimal location. The placement of the well should avoid problems such as sloping layers or impermeable layers which may result in a significant part of the contaminants being transported away from the monitor well.

One should be aware that construction of a well with a 0.25 m screen requires precise knowledge of the location of the groundwater table, since the screen otherwise will not be placed correctly. In practice, this means that the groundwater table must be measured in existing wells. A well with a screen length of only 0.25 m will be very vulnerable with respect to variations in the water table. Such a well can therefore only be counted to be useful for the calculation of the source term at the time of construction, and will most often not be useful for subsequent monitoring.

Screens with an interval greater than 0.25 m can be used to measure the resulting contaminant concentration in the top of the aquifer if sampling takes place with a very low yield so that no significant cone of depression is formed.

If a screen with an interval of greater than 0.25 m is used, the resulting contaminant concentration C_1 in the aquifers upper 0.25 m shall be calculated by:

$$C_1 = C_{1,\ meas} \bullet l/0.25\ m$$

where $C_{1,\ meas}$ is the measured contamination concentration $[ML^{-3}]$
and l is the effective screened interval (measured in metres).

In the formula for calculating the resultant contaminant concentration C_1, a constant concentration ,C_0, at the source of contamination is assumed for the entire contaminated area.
If it is justified by the investigation phase, the contaminated area can be divided into areas which each have a different area-weighted source term concentration.

For contamination over large areas, calculations can concentrate on contamination in a central area, cf. the principle that the zone with the highest concentration in the groundwater must comply with the groundwater quality criteria.

All parameters in the calculation participate linearly, i.e. uncertainties in the value of one parameter has the same impact as uncertainties in another parameter.

The greatest uncertainty will often be in determining the hydraulic conductivity and the source term concentration.

There will be some uncertainty in defining net infiltration (cf. Appendix 5.8, regional values for net precipitation are often used), and the hydraulic gradient (which can be read on regional potentiometric surface maps, or which can be determined during the investigation phase, in which case the interpolation method between points (linear interpolation, etc.) affects the result).

The least uncertainty is often connected with determination of the area and breadth of the contaminated area.

5.4.2.2 Step 2. Downgradient mixing model:

Description

The conservative calculation which is used, assumes that the pore water at the bottom of the unsaturated zone has a concentration of contaminants which is equal to the concentration at source. After this, the calculation assumes that the contamination is mixed in the upper part of the aquifer.

The resultant concentration of contaminants is calculated for a point at a distance from the pollution source which corresponds to one year of groundwater flow (calculated with the pore-water velocity of the groundwater); though maximum 100 m. At this theoretical calculation point, the groundwater quality criteria must be met.

In the saturated zone, a thickness of the mixing layer, d_m, is used (cf. Appendix 5.6):

$$d_m = (72/900 \cdot a_L \cdot v_p \cdot t)^{-0.5}$$

where

a_L is the longitudinal dispersivity [L],
v_p is the pore water velocity [LT^{-1}], and
t is the groundwater flow time; t has a maximum of 1 year as the theoretical calculation point is located at a distance which corresponds to 1 year′s flow distance.

If the actual thickness of the aquifer is less than 0.25 m, the actual thickness is used.

Calculations are carried out in a manner analogous to calculations in Step 1 of the risk assessment. The concentration of contaminants, C_2, in the upper part of the aquifer will be (cf. Appendix 5.6):

$$C_2 = \frac{A \cdot N \cdot C_0 + B \cdot d_m \cdot i \cdot C_g}{A \cdot N + B \cdot d_m \cdot k \cdot i}$$

C_0 is the concentration at the source [ML^{-3}],
C_g is the natural background content of contaminants in the groundwater [ML^{-3}],
A is the size of the contaminated area [L^2],
N is the net infiltration [LT^{-1}],
B is the breadth of the contaminated area (calculated in relation to the flow direction) [L],
d_m is the mixture depth in the aquifer at a given distance [L],
k is the aquifer’s hydraulic conductivity [LT^{-1}], and
i is the hydraulic gradient (unitless).

If the contaminant concentration C_1 in the upper 0.25 m of the groundwater aquifer in the near-source model (step 1) is measured, and subsequently corrected for screen length (if necessary), this can be used for a simple calculation of the downgradient contaminant concentration, C_2.

In the actual point of calculation, the resulting contaminant concentration C_2 can be expressed by:

$$C_2 = C_1 \bullet (0.25 \text{ m}/d_m)$$

where

C_1 is the resulting contamination concentration [ML^{-3}] in the upper 0.25 m of the saturated zone at the contamination source,

d_m is the thickness of the mixing layer after 1 year's groundwater transport, though maximum 100 m downgradient of the contamination. If the thickness is less than 0.25 m, then d_m = 0.25 m is used.

5.4.2.3 Step 3. Downgradient risk assessment with dispersion, sorption, and degradation

In the third step of the risk assessment, the resultant concentration of contaminants in the groundwater is calculated taking dispersion, sorption and degradation into account. Step 3 is a continuation of step 2, since the starting point for step 3 is the resulting contaminant concentration, C_2, which was calculated in step 2.

The resultant concentration of contaminants is calculated as in Step 2 of the risk assessment, at a point located at a distance from the contamination source, which corresponds to the groundwater flow distance in one year (calculated with the pore water velocity of the groundwater); but up to a maximum of 100 m. At this theoretical calculation point, the groundwater quality criteria must be met.
As previously mentioned, the step 1 and 2 risk assessments are conservative. The step 3 risk assessment cannot be carried out strictly conservatively. Where degradation is used in the calculations, monitoring should be carried out.

It is assumed that the saturated zone is homogenous and isotropic with constant groundwater velocity. In the saturated zone, degradation and vertical dispersion are taken into account. Degradation is assumed to be described by first-order degradation. Calculations are carried out based on typical first-order degradation constants, which are not necessarily conservative.

In order to use a step 3 risk assessment, the local geology and hydrogeology must be so well-known that sampling and monitoring screens can be placed in optimal locations (vertically as well as horizontally) in the contaminant plume and on a flow path downgradient of the contamination source. In addition, it is assumed that the contamination investigation has shown that the redox conditions give a possibility for the degradation of the contaminants of interest. It is thus assumed that the contamination does not contain recalcitrant contaminants in concentrations above groundwater quality criteria (for example, MTBE in a petrol contamination) and that in the theoretical calculation point, concentrations of potential degradation products also meet groundwater quality criteria.

When the above assumptions are met, the risk assessment should be carried out using first order degradation constants, which are typical for the relevant redox zone. The starting point for the calculation is the resulting contaminant concentration which was calculated in step 2. If the risk assessment shows that the groundwater quality criterion is not met even though natural attenuation is taken into account, the contamination then presents a risk. A new risk assessment may be performed, if new site-specific data is obtained from supplemental investigations.
On the other hand, if the risk assessment shows that the groundwater quality criterion is met, monitoring shall be carried out in order to ensure that degradation and redox conditions are as expected. In addition, monitoring shall provide data which can be used as a basis for calculating a site-specific degradation constant.

On the basis of first-order degradation, the resulting contaminant concentration after degradation, C3, can be expressed by the following formula /56/:

$$C_3 = C_2 \bullet \exp(-k_1 \bullet t)$$

where

C_2 is the resulting contaminant concentration calculated in the downgradient mixing model (step 2) [ML^{-3}],
k_1 is the first-order degradation constant in the saturated zone [T^{-1}],
t is the time period within which degradation takes place [T].

Sorption can be taken into account in order to assess the length of the time period during which contaminants are liable to be degraded. This is done by assuming that contaminants are transported to the theoretical calculation point with a velocity V_s which is given by

$$V_s = V_p / R, \qquad R > 1$$

The effect of sorption is that the contaminants require more time (more than one year) to reach the theoretical calculation point, and that more time is therefore available for degradation to take place in.

The so-called retardation coefficient R is defined in appendix 5.6 Retardation coefficients are dependent on the soil’s bulk density, the soil’s content of organic matter and the octanol-water distribution coefficient.

An overview of the parameters used for the calculations in the various steps of a risk assessment are shown in figure 5.10.

Where degradation is taken into account, the first-order degradation constant and the degradation time are the most sensitive parameters in calculating the resulting contaminant concentration. These parameters participate exponentially in the calculation of the resulting contaminant concentration while precipitation and the measured contaminant concentration in the top 0.25 m of the aquifer, for example, are participate only linearly in the calculation. A small change in the first-order degradation constant (or degradation time) can therefore result in a large change in contaminant concentration.

Calculation parameter	Step where calculation parameter is used
Measured contaminant concentration in pore water	1a 1b 2a 2b
Relative volume of air, V_l	1b 2b
Relative volume of water, V_V (= water-saturated porosity e_W)	1b 2b 3
Relative volume of soil, V_j	1b 2b
Soil temperature, T	1b 2b
Particle density of soil, d	1b 2b

Concentration of contaminant in soil, C_T		1b				2b			
Soil density, ρ		1b				2b			3
Soil content of organic matter, f_{OC}		1b				2b			3
Partial pressure of contaminant, P		1b				2b			
Molecular weight of contaminant, m		1b				2b			
Gas constant, R		1b				2b			
Solubility of contaminant in water, S		1b	1c			2b	2c		
Octanol-water distribution coefficient, K_{OW}									3
Longitudinal dispersivity, α_L					2a	2b	2c	2d	
Net precipitation, N	1a	1b	1c		2a	2b	2c		
Hydraulic conductivity, k	1a	1b	1c		2a	2b	2c	2d	
Hydraulic gradient, i	1a	1b	1c		2a	2b	2c	2d	
Area of contamination, A	1a	1b	1c		2a	2b	2c		
Breadth of contaminated area, B	1a	1b	1c		2a	2b	2c		
Effective porosity, e_{eff}					2a	2b	2c	2d	
1st order degradation constant in the saturated zone, k_1									3
Natural background content of contaminant in groundwater, C_g	1a	1b	1c		2a	2b	2c		
Measured concentration of contaminant at the top of the aquifer, $C_{1,\ meas}$				1d				2d	
Effective screen length, l				1d				2d	

Figure 5.10
Calculation parameters involved in Step 1-3 of the risk assessment. The letters a-d refer to the different methods through which the contaminant concentration at the top of the aquifer is determined.

a) Determination of the source term concentration using measured concentrations in pore water.
b) Source term concentration determined by the fugacity principle in porewater.
c) Source term concentration is set equal to the solubility.
d) Measurement of contaminant concentration at the top of the aquifer.

Redox processes are decisive for whether a substance degrades and how it degrades /13, 56/. The amount of energy obtained from various redox processes are vary different. Therefore, a thermodynamic order of processes can be set up reaching from aerobe processes which have the greatest energy output, through a variety of anaerobe processes, and ending with methanogene processes which have the smallest energy output.

In the saturated zone, development of redox sequences is often observed downgradient of a severe contamination. The most reducing conditions (for example, methanogene conditions) are found closest to the source of contamination where the concentration of organic matter is the greatest, whereas oxidising conditions (aerobe or nitrate-reducing conditions) are found at the edge of the contaminant plume, where the concentration of organic matter is low.

In order to chose the relevant degradation constant, it is therefore important to be aware of the redox conditions.

Aerobe conditions are present if the O_2 concentration in the groundwater is greater than 1 mg/l. Definitions of the various anaerobe redox zones, however, is more ambiguous. The anaerobe redox zones are in general characterised by the following:

- there is an increase in the hydrogen content (H_2) throughout the entire anaerobe zone
- in the nitrate-reducing zone, the concentration of nitrate (NO_3^-) decreases
- in the iron-reducing zone, the concentration of Fe(III) decreases and the content of Fe(II) increases
- in the sulfate-reducing zone, the sulfate content (SO_4^{2-}) decreases
- in the methanogene zone, methane (CH_4) is produced.

In a technology project, the Danish EPA has tabled first-order degradation constants, which are judged to be representative for Danish conditions /56/.

Often, few or no degradation constants can be found for specific redox conditions. Where publicised constants are found, these tend to vary greatly in value.

BTEXs, some chlorinated solvents and phenol are the only contaminants where there is a useful set of data making it possible to identify typical first-order degradation constants. The order of magnitude of degradation constants for these contaminants is found in appendix 5.8.

As mentioned earlier, step 3 of the risk assessment is not strictly conservative. Where the risk assessment shows that natural attenuation of the contaminants in groundwater can cause the groundwater quality criterion can be met, monitoring shall take place to ensure that the calculated concentrations in the monitoring wells is not exceeded. In addition, the actual redox conditions at the site shall be determined and data which can be used as a basis for calculating a site-specific degradation constant shall be obtained.

In connection with this monitoring, it is of greatest importance that the extent of the contamination and the groundwater flow is thoroughly mapped since monitoring will take place in wells which are located downgradient of the contamination along a flow line.

The exact placement of a monitoring point, the number of monitoring points analytical parameters and monitoring period depend on the contaminants composition and the hydrological conditions and must therefore be assessed in each specific case. Investigation wells may in some cases be used for monitoring purposes.

Methods for documenting that degradation actually takes place is given in the literature /56, 57, 58/. To carry out these methods, a number of chemical analyses are need as a supplement to the typical data which is used in cases where the risk assessment is simply based on mixing. Content of the contaminants and possibly of degradation products must be analysed for. In addition, analyses and field measurements which describe the redox conditions must be carried out.

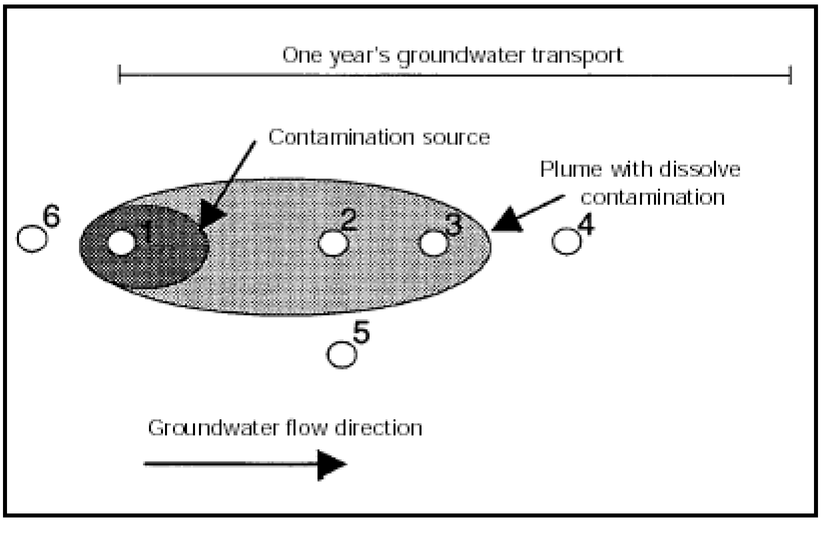
An example of how degradation of benzene can be determined is given in appendix 5.7.

As a minimum, three wells along a flow line (in addition to wells which are used to map the flow direction and the extent of contamination) are required, see figure 5.11. As a minimum, monitoring will typically be carried out twice a year for three years.

It should be noted that the monitoring wells should be placed relatively close to the source of contamination (at a distance of less than one year's groundwater transport), where there is the greatest possibility of an optimal placement in relation to the contaminant plume.

If concentrations exceeding the groundwater criterion are observed outside the calculated contaminant plume (upgradient or sideways) or at a greater downgradient distance than one year's groundwater transport, a new risk analysis based on additional data must be carried out, or one must conclude that the contamination presents a risk for the groundwater resource.

Figure 5.11



Example of the placement of investigations/monitoring wells in relation to a contamination /56/.

If degradation takes place more slowly than expected, a new risk assessment with the actual site-specific degradation constant shall be carried out.

When the actual site-specific degradation rate is to be determined from the monitoring data, measured contaminant concentrations must be corrected for the effects of sorption, dispersion and dilution. This can be done by comparing the concentration of the contaminant of interest with the concentration of a conservative compound (tracer) or by comparing a slowly-degradable compound in the contaminant with the easily-degradable contaminant of interest.

One easy method is to use a non-degradable compound in the contaminant as a tracer. If a slowly-degradable compound is used as a tracer rather than a non-degradable compound, a more conservative degradation constant results.

A description of how a non-degradable organic compound can be used as a tracer is found in appendix 5.6. An example of the use of an slowly-degradable organic compound as a tracer is found in appendix 5.7.

5.4.2.4 Numeric computer models:

There are numerous computer models available with varying calculation possibilities and bases for calculations. It is important to know the model’s basis for calculation (the equations on which calculations are based) as this can determine which assumptions must be used for the calculations.

One advantage of computer models is that they do not require that estimates be made, but are capable of performing numerical calculations. Also, such models can handle large volumes of data.

Numeric computer models can therefore perform calculations using many horizontal layers, and with horizontal variations in the properties of the material.

However, in order to benefit from the numeric models, large amounts of data are required. It is therefore often expensive and time-consuming to conduct risk assessments using computer models.

5.5 Surface-water recipients

Groundwater discharge and - in special circumstances - surface run-off of contaminated water can affect surface water recipients such as streams, lakes, and coastal areas.

Calculation of the adverse effect of contaminants on a recipient is based on the same principles mentioned in connection with groundwater contamination.

General guidelines for the quality of water in recipients are laid down in the Recipient Quality Plans, drawn up by county councils.

Based on the objectives for the recipients, county councils lay down specific limit values for discharges. County councils must ensure that, as a minimum, discharges comply with the relevant requirements in statutory orders and guidelines /47, 48, 49, 50, 51, 52/.

Variations in mobility and physical/chemical/biological conditions imply that, as a rule, the parameters which are critical for an aquifer are different from the parameters which are critical for a recipient.

For example, a small hydrocarbon spill may have no impact on a well-protected aquifer. On the other hand, a direct surface run-off of even small amounts of oil to a nearby pond will be unacceptable.

Surface-water run-off to a recipient can occur if:

- The recipient is located in the immediate proximity of the site.
- The topography and run-off conditions in the area make surface-water run-off possible

For contamination via normal surface-water run-off (during a rain storm or a sudden thaw), contamination will primarily comprise contaminant bound in the upper soil layer. Therefore, to assess this effect, results of analyses of topsoil samples are used.

With respect to surface water run-off, it is rarely possible to judge the size of the effect on the recipient.

The recipient can only be contaminated through groundwater discharge if groundwater flows from the site to the recipient. Recipients located at a higher level than the site are therefore without risk.

6 Quality criteria for soil, air and groundwater

6.1 Background and objectives

In recent years, the Danish EPA has recommended a number of criteria for selected chemical substances by issuing several publications and by background work /9, 10, 19, 20, 21, 22, 23, 24, 25/.

The objective of this chapter is to review the various criteria for soil, air, and groundwater. The basis for setting these criteria and their application will briefly be described.

The criteria for the three media – soil, water, and air – have been designed independently of each other in that they are based on exposure to each of the three media. Each criteria should thus be complied with simultaneously. Compliance with soil criteria, for example, does not automatically imply compliance with criteria for water and soil.

It should be emphasised that the soil and groundwater criteria do *not* represent a risk analysis. A specific risk analysis should take account of the local geological conditions and the sensitivity of land use.

6.2 Quality criteria for soil

The Danish EPA has prepared toxicological quality criteria for soil for a large number of selected chemical substances.

Criteria are always based on the assumption that it should be possible to use the site for very sensitive purposes (e.g. private gardens or day-care centres). As a starting point, soil-quality criteria should be complied with from the surface to a depth of 1 m, however, see also Section 5.2.3. In addition to health considerations, the criteria are also based on aesthetic and hygiene considerations (smell/appearance/taste).

As a supplement to, or instead of, soil-quality criteria, quality criteria for evaporation apply for several substances. Air-quality criteria are described in Section 6.6.

In most cases where contamination is to be assessed, ecotoxicological considerations are not taken into account. The current ecotoxicological quality criteria are listed in Table 6.1.

For most of the substances reviewed, quality criteria for the top soil are based on acceptable/tolerable daily exposures. If it is known that exposure to contamination components occurs from other sources (e.g. food or air), the criteria will take account of this and ensure that total exposure does not exceed the ADI/TDI values (acceptable daily intake / tolerable daily intake).

Table 6.1 lists quality criteria for soil in connection with very sensitive land use. To aid risk analysis (Section 5.1), the table also shows whether the reason for setting the quality criteria is the acute or the chronic effect of the substance. If individual values significantly exceed the soil quality criteria (+100%), the need for mapping the extent of the high value/hot spot should be assessed.

The statements of acute and chronic effects in Table 6.1 do not mean that a threshold criterion can be derived for a substance where the level is based on the chronic effect.

Table 6.1 also states zero values for selected organic components. Zero values are defined as calculated pore-water concentrations for clay and sand (Table 1 in Appendix 5.3) which do not exceed the quality criteria for the groundwater.

Table 6.1
Quality criteria for soil in areas with very sensitive land use, ecotoxicological quality criteria and background levels /3/. The methods of analysis for individual substances appear in the guidelines on sample-taking and analyses /3/. All units are in mg/kg dry weight (DW).
Note: For several substances (marked *), criteria for evaporation are stated (see Table 6.5).
[1] Based on acute effect.
[2] Based on chronic effect.
[3] PAH, total defined as the sum of individual components: fluoranthene, benzyl(b+j+k)fluoranthene, benzyl(a)pyrene, dibenzyl(a,h)anthracene, and ideno(1,2,3-cd)pyrene.
[4] See also /53/.
[5] For hydrocarbons greater than C_{35}, see /3/.

Substance	Soil quality criteria	Eco-toxicological soil quality criteria[4]	Background level
Acetone	8		
Arsenic	20[1(2)]	10	2-6
Benzene	1.5[*2]		
BTEX, total	10[*2]		
Cadmium	0.5[2]	0.3	0.03 – 0.5
Chloroform	50[*2]		
Chlorophenols, total Pentachlorophenol	3[*2] 0.15[*]	0.01 0.005	
Chromium, total Chromium (VI)	500 20	50 2	1.3 – 23
Copper	500[1]	30	13
Cyanide, total Cyanide, acid volatile	500 10[*2]		
DDT	1		
Detergents, anionic	1,500[2]	5	
1,2-dibromomethane	0.02[2]		
1,2-dichloroethane	1.4[2]		
1,1-dichloroethylene	5[2]		
1,2-dichloroethylene	85[2]		
Dichloromethane	8[2]		
Fluorides, inorganic	20[1]		

Gas oil Total hydrocarbons ($C_5 – C_{35}$)[5]	100		
Lead	40[2]	50	10 – 40
Mercury	1	0.1	0.04 – 0.12
Molybdenum	5	2.0	
MTBE	500[2]		
Nickel	30[1]	10	0.1 - 50
Nitrophenols Mono- Di- Tri-	 125[2] 10[2] 30[2]		
PAH, total Benzo(a)pyrene Dibenzo(a,h)anthracene	1.5[2,3] 0.1[2] 0.1[2]	1.0 0.1	
Petrol (C_5-C_{10})	25[*]		
Petrol (C_9-C_{16})	25[*]		
Phenols, total	70[*1]		
Phthalates, total DEHP	250[2] 25[2]	 1.0	
Styrene	40[*2]		
Turpentine, mineral ($C_7 – C_{12}$)	25[*]		
Tetrachloroethylene	5[*2]		
Tetrachloromethane	5[*2]		
1,1,1-trichloroethane	200[2]		
Trichloroethylene	5[*2]		
Substance	Soil quality criteria	Eco-toxicological soil quality criteria[4]	Background level
Vinyl chloride	0.4[*2]		
Zinc	500	100	10 - 300

6.3 Cut-off criteria for soil

In the guidelines from the Danish EPA on ‘Advice to Residents in Slightly Contaminated Areas’, cut-off criteria for soil contamination in the upper soil stratum have been set for ten selected substances /4/.

The cut-off criteria state the level of soil contamination at which it is necessary to completely cut off all contact with the soil, for example by remediation or excavation, if the area is to used for very sensitive purposes.

The Guidelines include a number of rules which aim at reducing exposure to the same level as for areas where the soil-quality criteria are not exceeded. The cut-off criteria for soil are listed in Table 6.2.

Table 6.2
Criteria for necessary contamination cut off, mg/kg dry weight (DW).
[1]: Based on acute harmful effects
[2]: Based on chronic harmful effects

Substance	Level where contamination cut off is necessary
Arsenic	20[1]
Cadmium	5[2]
Chromium	1,000
Copper	500[1]
Lead	400[2]
Mercury	3
Nickel	30[1]
Zinc	1,000
PAHs	15[2]
Benzo(a)pyrene	1[2]
Dibenzo(a,h)anthracene	1[2]

If contamination in the upper soil stratum is not more than the cut-off criteria, and there are no clumps or hot spots with high concentrations of contamination, then the completed surveys (the initial (historical) survey and analyses of soil samples) provide evidence that the criteria described in these guidelines have been complied with /4/.

As it can not be ruled out that contamination near the surface may represent a threat to the groundwater, these criteria can be supplemented with other criteria or specific risk assessments, cf. Section 5.4. Furthermore, contamination should not contain substances which are problematic because they evaporate into the indoor and/or outdoor air, cf. Section 5.3.

If a larger area is slightly contaminated, i.e. concentrations are less than the cut-off level, situations may arise where remediation is carried out at lower concentrations, for example the soil-quality criteria. For example, this may occur in situations where an enforcement notice has been issued or where limitations on land use following remediation, are undesirable (corresponding to de-listing a contaminated site).

6.4 Criteria for soil from other sources

A distinction is made between acceptable residual concentrations in soil which is brought in from other sources and acceptance criteria for untouched soil.

Soil which is filled in an excavation at a contaminated site, for example, can originate from other parts of a site, from off-site, or from soil remediation. With regard to soil from other sources, land use or the depth at which the soil is to be laid is not considered.

Therefore, the overall requirement is that soil from other sources is clean, or has previously been cleaned to the extent that soil-quality criteria are complied with to the bottom of an excavation.

It should be emphasised, that mixing contaminated soil with clean soil with a view to complying with acceptance criteria is in direct conflict with the intentions of environmental legislation. Dilution in this way is unacceptable, irrespective of whether it arises in required excavations or through mixing soil from the same site.

Use of soil from other sources must always be approved separately for each specific case.

6.5 Quality criteria for groundwater

Table 6.4 shows quality criteria for groundwater. The quality criteria stated in the Table correspond to 'requirements for groundwater for normal aeration and filtration' in the 'Guidelines on Well Control at Water Works' /26/. These guidelines also include requirements for a number of other substances and parameters.

In connection with the preparation of health-based quality criteria, a data sheet has been prepared for a large number of the substances mentioned in Table 6.4, where the most important data and conclusions from the background documentation are described. These data sheets have been collected in a report called 'Toxicological Quality Criteria for Soil and Drinking Water' /9/.

Table 6.4
Quality criteria for groundwater beneath contaminated sites. 1)Sum of fluoranthene, benzo(b)fluoranthene, benzo(k)fluoranthene, benzo(a)pyrene, benzo(g,h,i)perylene, indeno(1,2,3-cd)pyrene

Substance	Groundwater quality criteria, µg/l	Background level, µg/l
Acetone	10	
Arsenic	8	0.1->8
Benzene	1	
Boron	300	10->300
Butylacetates	10	
Cadmium	0.5	0.005->0.5
Chlorinated solvents (not vinyl chloride)	1	
Chloroform	As low as possible	
Chromium, total	25	0.04-10
Chromium VI	1	
Copper	100	0.1->50
Cyanide, total	50	
DEHP	1	
Detergents, anionic	100	
1,2-dibromomethane	0.01	
Diethylether	10	
Isopropyl alcohol	10	
PAH 1)	0.2	
Lead	1	0.1->1
Methylisobutylketone	10	
Methyl-tert-butylether (MTBE)	30	
Mineral oil, total	9	
Molybdenum	20	0.2-20
Naphthalene	1	
Nickel	10	0.1->10
Nitrophenols	0.5	
Pentachlorophenol	Not measurable	
Pesticides, total Pesticides Pesticides, persistent chlorinated	0.5 0.1 0.03	
Phenols	0.5	
Substance	Groundwater quality criteria, :g/l	Background level, :g/l
Phthalates (not DEHP)	10	
Styrene	1	
Toluene	5	
Vinyl chloride	0.2	
Xylenes	5	
Zinc	100	0.5->10

The quality criteria for groundwater have been set on the basis of a total assessment of the water's route to the consumer. Groundwater which is to be used as drinking water must be of such a quality that, after all influences to which it is exposed before it reaches the consumer, the water still complies with the requirements for drinking water.

The physical-chemical conditions and the vulnerability of the groundwater aquifer determine the spread from the source to the groundwater aquifers, and thus influence the quality requirements.

Quality criteria must be fulfilled in the major groundwater aquifers. The quality requirements must also be fulfilled in minor, upper groundwater aquifers which may be the cause of significant spreading of contamination, or which may be used for drinking-water supplies.

An important stage of risk assessment (Chapter 5.4) is therefore to establish an acceptable level for residual contamination in relation to an influence on the groundwater at the site.

6.6 Quality criteria for air

In connection with assessing evaporation of highly volatile components from contaminated sites, there is a distinction between the terms ‘evaporation criteria’, and ‘limit values for air’. A definition of these terms appears in Table 6.5.

Table 6.5
Definitions of criteria for air

Evaporation criteria	An air-quality criterion for evaporation to air is a contribution value which is generally set equal to the limit value for air (see below).
Limit value for air	The Danish EPA lays down limit values for air on the basis of toxicological assessments. The limit values for air are used for setting maximum permissible contributions of contaminating industries of substances to the air as immission (B values). They are also used in setting air-quality criteria for evaporation into the overlying air.

A number of criteria for evaporation from soil have been laid down. These criteria can be applied in assessing non built-up areas, and in assessing evaporation to the indoor air in overlying buildings.

As a general rule, evaporation of chemical substances from a contaminated site should not give rise to a higher contribution to the overlying air than laid down in the evaporation criteria. Air-quality criteria for evaporation to the overlying air generally correspond to the limit values for air issued by the Danish EPA.

Table 6.6 lists the quality criteria for evaporation to the overlying air.

Table 6.6
Quality criteria for evaporation from contaminated sites.

Substance	Air-quality criteria for evaporation to the overlying air mg/m^3
Acetone	0.4
Aromatic hydrocarbons $C_9 – C_{10}$, total	0.03
Benzene	0.000125
Butylacetates	0.1
Chloroform	0.02
Cyanides, volatile acidic	0.06
Diethylether	1
Isopropanol	1
Hydrocarbons, total	0.1
Methylisobutylketone	0.2
MTBE	0.03
Naphthalene	0.04
Phenols Phenol Methylphenols (cresols) Dimethylphenols (xylenols)	 0.02 0.0001 0.001
Chlorophenols, sum of chlorophenols, dichlorophenols, trichlorophenols, and tetrachlorophenols Pentachlorophenol, PCP	0.00002 0.000001
Nitrophenols	0.005
Styrene	0.1
Tetrachloromethane	0.005
Tetrachloroethylene	0.00025
Toluene	0.4
Trichloroethylene	0.001
Vinyl chloride	0.00005
Xylenes	0.1

Limit values for air have been established for many more substances than mentioned in Table 6.6. These values can be extracted from the background documentation to the list of B values cf. the four substance categories below. In addition, the Danish EPA regularly sets limit values for air for other substances, and the values already listed are regularly revised.

When implementing limit values for air to B values, there are four different categories /19/.

- Substances where only the total dosage and thus, in reality, the average concentration of the substance, is decisive for an effect. In these cases the B value is set at 40 times the limit value for air.
- Substances with acute or sub-acute effects. In these cases B values are set at the limit value for air.
- Substances where smell is the limiting factor. In these cases B values are set at the limit value for air.
- Substances which have an immediate acute effect. Here the B value is set at 1/10 of the limit value for air.

Refer also to the background documentation on setting B values /21/.

6.7 Use and limitations of quality criteria

It should be noted that groundwater criteria are independent of soil criteria in that compliance with soil-quality criteria does not automatically ensure compliance with criteria for water.

Similar to groundwater criteria, evaporation criteria are also independent of soil criteria, in that compliance with soil criteria does not automatically imply compliance with evaporation criteria.

Of course, it would be very practical if it were possible to have overall criteria for soil which could also ensure that evaporation or leaching did not occur to a significant degree. At the moment, this is not possible.

Situations can arise where it is necessary to ignore quality criteria. For example, soil-quality criteria should not be applied if the natural background level for a given heavy metal is higher than the soil quality criteria. Similarly, there may be areas where the groundwater has a naturally elevated content of chloride or organic substances.

7 Reporting

It is important that results from investigations are reported in an easy-to-understand manner. Texts should be clear and concise, and the text should be supported by summary tables and figures.

7.1 Preliminary investigations

7.1.1 Report outline

An example of an outline for use in a report of a preliminary investigation is given below.

Abstract
Table of contents
Table of appendices

1. Introduction
2. Site description
    2.2 Historical review
    2.3 Current and future land use
    2.4 Water abstraction and nearby surface water recipients
3. Objectives and strategy
4. Scope of the investigation
5. Geology and hydrogeology
6. Extent of contamination
    6.1 Contamination of soil
    6.2 Contamination of groundwater
7. Risk assessments
8. Summary and recommendations
9. References

The report should start with an abstract. This should provide a brief and accurate preliminary impression of the content of the report. Details such as datum and exact concentrations of contamination are not usually included in the abstract.

The report should be divided into main sections and subsections which should be numbered using Arabic numerals. More than three levels should be avoided for the sake of simplicity. If the need arises, an additional main section is preferable to additional levels.

The first part of the report should also include a table of appendices which are included in the report.

The introduction should contain the following information on the site and the investigation:

- Address
- Number from the parcel register

- Contaminated site number, if any, or other registration number
- Owner
- Party who ordered the investigation (client)
- Drilling contractor and analytical laboratory
- Consultants, if relevant
- Short description of the situation

Chapter 2 on the description of the site may contain the sections mentioned in the outline:

- Historical review
- Current and future usage of the site
- Water abstraction and nearby surface-water recipients

The objective of the investigation should be stated, and the strategy chosen should be described. The reasons for selecting the strategy should be given in light of the site description and the investigation objectives.

Chapter 4 describes the scope of the investigations conducted. Technical details, such as drilling work, sampling, methodologies for geophysical investigations and the like may be referred to the appendices.

The programme chosen for analysis can be stated in the text, possibly in the form of a table. If the programme is extensive, it may be preferable to attach it as an appendix.

The regional geology and hydrogeology should be described on the basis of available literature and geological cyclogram maps (which show lithographic strata and groundwater levels and are prepared from data from soil borings). Relevant geological cross sections should be prepared as well as maps of the potentiometric surface for relevant aquifers. The vulnerability of the aquifer should be assessed on the basis of the geological conditions and the potentiometric maps.

On the basis of the geology and hydrogeology close to the surface, the risk of affecting the quality of surface water recipients should be assessed.

Furthermore, the risk of affecting neighbouring sites should be assessed.

This chapter provides a description of contamination type, concentration, and extent in relation to the quality criteria for the relevant medium.

The results of chemical analyses are attached as laboratory reports in appendices. It may be appropriate to collect results in the form of a table within the text, or, if the results are very extensive, in an appendix.

For a very large number of results, presenting them in table form alone may not provide the necessary overview. In many cases, it may be appropriate to present results graphically.

Chapter 7 should collate the geological and hydrogeological information with information on the extent of the contamination. On this basis, an assessment of the contamination conditions on and around the site being investigated is presented. A risk assessment should be carried out with a view to providing recommendations regarding possible supplemental investigations or remedial measures.

A summary of the investigation should be provided, as well as the most important results.

7.1.2 Figures and tables

Figures and tables are a very important part of a report. They are included to facilitate understanding and to provide an overview. If more than a couple of numbers are to be presented in a text, they are more easily comprehended in a table.

Appendix 7.1 describes the figures which a preliminary investigation report may contain. In many cases, there will be insufficient space for data to be made satisfactorily clear in a figure in the text. In this event, they should be placed in an appendix.

7.1.3 Appendices

Appendix 7.1 provides a description of the appendices and figures a report may contain. The description may be supplemented by further appendices as required.

If there are very extensive appendices, it may be appropriate to include these in a separate report.

7.2 Supplemental investigations

In principle, reports of supplemental investigations should designed around the same outline as reports of preliminary investigations.

However, a general outline for all supplemental investigations cannot be given, as the investigations may have varying objectives and content.

It should be mentioned that relevant results from previous investigations should be included and worked into reports of supplemental investigations.

The overall assessment of results will often appear as a risk assessment. If there is a risk, investigations will often result in preparation of a proposal for an outline project for remediation.

8 Design

8.1 General conditions

This chapter deals with the situation where remedial measures are to be carried out. The Danish EPA has published a 'Project Management Handbook for Soil and Groundwater Contamination' (1995) /27/ (*Projektstyringshåndbog for jord- og grundvandsforureninger*), to which this chapter refers.

For decision-making, it is important that an organisation chart and clear divisions of responsibility are established prior to initiating a remediation project. Areas of responsibility and authority for specialist work, deadlines, and costs, should be made clear in the organisation of the project.

In addition to the Environmental Protection Act and the Act on Waste Deposits, a number of rules and guidelines will be used in remedial projects, for example sections from the Health and Safety at Work Act, building regulations, rules for plant and technical installations, instructions, etc. The most important rules appear in the 'Project Management Handbook' /27/.

Permits should be obtained from local authorities and counties. Amongst other things, permits are required from counties for:

- Discharge into the rainwater system and recipients
- Abstraction of groundwater (for pump-and-treat purposes)
- Establishing treatment plants
- Establishing landfills
- Constructing fences (non-urban zones)

Permits are required from local authorities for:

- Emissions into the air (Chapter 5 enterprises)
- Discharge to sewers
- Excavation work
- Demolition of buildings
- Disconnecting sewers
- Constructing fences (urban zones)

The local authority must also assign landfills / treatment facilities for waste and contaminated soil.

If a change in land use or a new construction is to take place on a contaminated site, there may be a benefit in combining the remedial project with the building project. In these cases, construction should be adapted to the contamination, so that it causes the least possible nuisance; for example for the new use (indoor air, outdoor area) and the work environment during construction work. Construction should be located such that it is possible to carry out remedial measures on any remaining contamination at a later date.

When risk analyses lead to implementation of remedial measures, the investigation phase should conclude with a 'standardised' project plan including preparation of an outline project, a detailed project, and subsequently, tender documents for the contractor. The relationship between sub elements in the various projects appears in Table 8.1.

Table 8.1
Relationship between outline project, detailed project, and tender documents for contractors.

	Outline project	Detailed project	Tender Documents
Technical details	Overall description of principle technical solutions	Detailed technical description	Detailed technical description*)
Cost	Rough costs	Detailed costs	Tender lists (TAG**) and TBL***))
Time and conditions	Overall time-table	Detailed timetable	AB92****) and Special conditions (SB)

**) In tender documents this is often referred to as 'description of work' or 'special description of work' (SAB) and 'drawings'.*
***) TAG is an abbreviation for tender and payment terms (tilbud og afregningsgrundlag).*
****) TBL is an abbreviation for tender lists (tilbudsliste),*
*****) AB92 is an abbreviation of standard terms issued in 1992 (Almindelige betingelser udgivet i 1992)*

In some cases it can be appropriate to conduct investigations and remedial measures using an approach based on normal building and construction projects /2/.

The EU has laid down rules for purchasing goods and services, and performance of construction work through EU tenders. The following threshold values apply in determining whether the product/service shall be put out to EU tender (in 1997 prices):

Type of contract	SDR *)	ECU	DKK
State service contracts	130,000	137,537	1,031,998
County and local authority service contracts a)	200,000	211,595	1,587,689
State purchase contracts	130,000	137,537	1,031,998
County and local authority purchase contracts	200,000	211,595	1,587,689
Public building and construction contracts and supply enterprises a)	5,000,000	5,289,883	39,692,229

*) SDR : Special Drawing Rights
a) In addition to these values, certain exemptions and supplements apply /55/.

Furthermore, it is necessary to note that there are a number of time limits for EU tenders.

8.2 The Outline Project

The outline project comprises preparation of one or several alternative projects which fulfil the remediation requirements. The following contents may be used for preparing the outline project:

1. Introduction
2. Background for the remediation project
3. Objectives, including extent of contamination and risk
4. Remedial measures
5. Operation and evaluation phase
6. Timetable and costs
7. Recommendations

The introduction should contain the following information about the site:

- Address, parcel register number, and contaminated site number, if any (location of site should be shown on a map)
- Site owner
- References to relevant investigation reports
- Party who ordered the work (client)
- Consultant

The background for implementing the remedial measures should be described on the basis of an historical review of the site and any previous investigations. Information about land use, groundwater interests and/or objectives for recipient quality, extent of contamination, and risk assessments etc. are relevant here.

The objective of implementing remedial measures and the requirements to be fulfilled by the remedial measures should be described under the objectives section.

One or more proposals for remedial measures which fulfil the above requirements should be given. For each proposal, a brief description should be given of:

- Criteria for the choice of technique
- Project description
- Tests necessary for design of the project
- Overview of the necessary permits, approvals etc. from the authorities
- Assessment of the environmental impact of the proposed measures
- Working-environment assessments of the proposed measures, taking account of the type of contamination

When considering alternative proposals for remedial measures, the costs, quality (environmental effect), and environmental impact (green accounting) of the individual proposals will comprise sub-elements. Comparative financial effect calculations may be included with a view to ensuring the greatest possible environmental and health results from the funds invested. Assessment of alternative remedial measures may be based on construction costs, operating costs, speed and effect of remediation, as well as secondary, derived environmental effects. In this way the best results are achieved for the funds invested. This assumes that the proposed remedial measures all remove the risks demonstrated.

A brief description of the operation and evaluation phase should be given for each solution proposal, including descriptions of any monitoring programmes.

A timetable for remedial measures should be provided, as well as a timetable for a subsequent operation and evaluation phase.

A cost estimate of total expenditures for the remediation project should be prepared. The estimate should include construction expenses as well as operating and evaluation expenses. Furthermore, the cost of preparation of the detailed project, tender documents, supervision, and reporting management for the remedial measures should be estimated. A standard statement of the various items should be prepared in outline and in detail so that costs can more easily be followed throughout the course of the project.

On the basis of the alternative proposals, recommendations should be given for the most appropriate remedial measures at the site.

8.3 The detailed project

The objective of the detailed project is to prepare a detailed description of the execution of the remedial measures. The detailed project should include a detailed technical description, a detailed timetable, a detailed cost overview, and a description of the operation and evaluation phase. Parts of the tender documents which are used in the tendering phase will also be included in the detailed project.

After the remedial project is selected during the preparation of the outline project, it is sometimes necessary to carry out supplemental investigations during detailed planning. These investigations are carried out in order to optimise and design the remediation. Examples include investigations of foundation conditions and detailed mapping of contamination. If in-situ remediation has been chosen, it will usually be appropriate to conduct in-situ tests, e.g. pump tests.

The introduction of the detailed project often includes information about the organisation of the project, location, etc., which is also part of the ‘Special Terms’ in the tender documents (see Section 8.4.2, and Appendix 8.1).

The technical description comprises a detailed description of what is to be carried out. It is often beneficial to prepare this description as a ‘Description of Work’, which is included in the tender documents, prepared prior to carrying out the project (see Section 8.4.2, and Appendix 8.1).

A detailed timetable for carrying out the remedial project should be prepared, as well as a timetable for a possible subsequent operation and evaluation phase.

Costs associated with the project should be broken down into expenditure on consultants, contractors, soil treatment/disposal, water treatment, analyses for documentation, working-environment measures, insurance, unforeseen expenses, operation and evaluation phase expenditure, etc. Individual items should be comparable with items in the outline project.

8.4 Invitations to tender and tender documents

8.4.1 Tender and contracting

A maximum of two tenders should be obtained for *closed tenders.*

If *more than one closed tender* is received, the following rules apply:

- If the tender is *cancelled*, a period of three months after the cancellation must expire before invitations for closed tenders for the same work or order can be made. However, invitations for open tenders may be held immediately.
- If a *closed tender is not cancelled*, a period of six months from receipt of the most recent closed tender must expire before an invitation for closed tenders may be held for the same work or order, and a new invitation for open tenders may not be held.

For invitations for open tenders, tenders are normally obtained from at least three bidders. If offers are obtained, the following rules apply:

- If the tender is *cancelled*, a period of three months after the cancellation must expire before invitations for closed tenders for the same work or order can be made. However, invitations for open tenders may be held immediately.
- If the *tender is not cancelled*, a period of six months from receipt of the tender must expire before invitations for closed tender may be held for the same work or order, and a new invitation for open tenders may not be held.

Cancellation must be reported in writing to each tenderer. See also the Invitations to Tender Act /28/.

Consultants may be chosen as a result of open or closed tenders. In this case, standard contracts between the client and the consultant are prepared. An example of such a contract is included in the Project on Soil and Groundwater from the Danish EPA No. 5, 1995 /27/.

In principle, there are three types of contracts in building and construction projects:

- Specialist contracts
- Main contracts
- Turnkey contracts

Specialist contracts correspond to the client entering into contracts with different contractors according to the division of specialist tasks. Contracts are therefore entered into with specialist firms with differing expertise. Use of this kind of contract requires careful organisation and management from the client or the consultant, as the different parts must operate both individually and together.

In main contracts, the developer enters into a contract with one contractor. After this, the contractor manages the contract which often involves sub-contractors. The developer only has an agreement with the main contractor which is then responsible for services and deadlines.

Turnkey contracts apply when the main contractor is responsible for implementation as well as planning/design of the work. This type of contract is rarely used for remediation projects.

8.4.2 Tender documents

Tender documents should contain a detailed description of the remediation project. An outline for the tender documents is given below:

- Letter of tender
- Contractor summary
- Standard terms
- Special terms (SB)
- Description of work
- Special description of work
- Tender and payment terms (TAG)
- Tender lists (TBL)
- Drawings
- Appendices

The individual parts of the tender material are described in more detail in Appendix 8.1.

The tenders received should be assessed the technical solution and for costs. Sometimes, decisions are based on an evaluation model where both costs and the technical proposal are assigned a specific weight. For example, an equal weighting may be assigned between the proposal and the costs. After choosing the project, the contract with the contractor should be prepared. An example of a contract is included in the ‘Handbook for Project Management of Soil and Groundwater Contamination’ /27/.

If the tender is to include a project proposal, the letter of tender should state that the project proposal will be included in evaluation of the tender, and that the client retains the right to accept the best project proposal, irrespective of the price.

8.5 Supervision

All remedial measures for contaminated soil or groundwater should take place under environmental and specialist supervision. The objective of supervision is primarily to ensure that remedial measures are implemented as described in the detailed project, and that remedial solutions operate in the best possible manner. Therefore, environmental supervision should be conducted with regard to the environmental effect of the remedial measures, and specialist supervision should examine the services provided by the contractor. The following tasks are usually performed as part of supervision:

- Ensure compliance with quality criteria (e.g. in field measurements and analyses)
- Ensure compliance with procedures for the working environment (e.g. in working-environment measurements)
- Contact with the authorities
- Concluding documentation for work completed (reports to the authorities)

It can often be beneficial to describe environmental supervision tasks in an action plan. Experience shows, for example, that excavation projects sometimes reveal contaminated areas which were not identified in previous investigations. Environmental supervision must ensure that such situations are handled according to the specifications in the detailed project.

A supervision report should be prepared as part of the supervision which documents that the quality standards for the project have been complied with. The report should contain a description of the performance of the remedial project. For example, for soil excavation, the report may contain descriptions of where soil has been excavated, the amount of soil which has been excavated, and the fate of the excavated soil. The report should also contain a properly scaled plan with the location of the samples collected for documentation purposes, excavation contours and analysis results, as well as documentation of the location and concentration of possible residual contamination.

Deviations from the original project should also appear. A more detailed description of supervision tasks appears in the 'Handbook for Project Management' /27/.

8.6 The work environment and the external environment

8.6.1 The work environment

In construction work on remedial measures, both client and the contractor are liable under the Health and Safety at Work Act. The client should prepare the necessary requirements for the contractor based on the Health and Safety at Work Act and experience. The proposal should be discussed with the Working Environment Authority, so that any problems or uncertainties can be resolved before work commences. It should be noted that the Working Environment Authority does not issue formal approval of work environment measures. The client and the supervision carried out by the client are responsible for ensuring compliance with the Health and Safety at Work Act. Written instructions for the supervisors on work environment measures should be prepared.

The contractor should be informed of the type, concentration, and extent of the contamination, as determined by investigations already completed. The tender documents should therefore include summaries of the results of previous investigations. In addition, there should be information on the requirements for work environment measures, such as choice of appropriate respiratory and other protective devices, personal hygiene, temporary site buildings and facilities, special conditions regarding movement to and from the work site (wash places, sluice systems etc.), technical measures such as use of pressurised cabs on excavators, rules for handling, transport, and disposal of contaminated soil. The Occupational Health Service for construction workers (BST) can also provide advice to the contractor before measures are implemented.

Prior to commencement of work, a plan for health and safety at the construction site is prepared /29/. The plan should be regularly updated and it should be accessible to employees and employers at the workplace throughout the construction period. Usually, the contractor conducting the construction work at the site is responsible for preparation of the health and safety plan. If two or more employers employ more than 10 people at the site at the same time, client is responsible for preparing and updating the plan. The plan should be discussed at a start-up meeting between the supervisors, contractor, Working Environment Authority, and the client.

A notification of the construction site must be made to the Working Environment Authority if the duration of work is expected to exceed 30 working days, and at least 20 workers are employed at the same time, or if the total work is expected to exceed 500 man days. The notification site should be made by the client. It is recommended that the Working Environment Authority is always notified of work on contaminated soil.

References to the Health and Safety at Work Act, statutory orders, and guidelines are in /27/.

8.6.2 The external environment

Planning should also include rules concerning the environment outside of the site. The following problems may arise, with suggested mitigating measures:

- Dust problems from soil work (dust measurements and spraying with water).
- Noise problems (noise measurements and restrictions to specific times)
- Odour problems (odour assessments, and covers or ventilation using air treatment filters.

Close contact with landowners and any residents is crucial to ensuring that a project progresses smoothly. This contact may include delivering reports, holding meetings before the project is put out to tender (gives the possibility for changes), agreements concerning restoration, information to neighbours (possibly via residents meetings), possible meetings during the project (for larger projects), and joint reviews after restoration.

8.7 Project and quality control

In order to ensure optimal control of remedial projects with regard to quality, time, and costs, it is appropriate to include project management, quality assurance, and environmental management.

Aids to project management include descriptions of activities, divided into sub-activities, organisation charts with descriptions of qualifications and tasks, timetables, progress control, budgetary controls, and distribution of documents.

Quality assurance should be carried out and include planning of controls, the actual control, and documentation of controls. The scope of quality assurance depends on the size of the project, but a QA plan should be prepared which, as a minimum, contains the sub-activities which are to be controlled, and the persons who are to perform such controls (should not be the same persons who perform the activities).

Quality assurance will also usually include document management and control, and project scrutiny.

The environmental management system is a relatively new tool which has won recognition as a control tool in remedial projects to ensure environmentally-correct execution and minimal environmental impact. The control elements in environmental management include the project's energy resources (consumption of energy, transport, and choice of machinery), consumption of water resources, lay out and operation of the building site, health effects for the manufacturers, users and neighbours, global, regional and local health effects, as well as technological-development considerations. The size of the project will influence the need for environmental management.

8.8 Completion of project

At the completion of work, handing over takes place according to AB92, paragraphs 28 and 29 /30/.

According to AB92, paragraph 37, the main contractor has a duty and right to assist in correcting outstandings which come to light after the hand over. /30/.

According to AB92, paragraph 37, the client must call a review of the work no later than one year after the hand over /30/.

According to AB92, paragraph 38, the client or main contractor may call for an inspection no later than 30 days before five years have expired after the hand over /30/.

When all criteria have been complied with, operations are concluded and a completion report is prepared.

A completion report should be prepared which documents compliance with the completion criteria. The report should primarily include observations made at the inspection.

9 Remedial Measures

This chapter is structured so that the ‘contamination media’ represent the three main sections (soil contamination, groundwater contamination, soil gas/gas contamination). As far as possible, the main sections are subdivided into remediation techniques for the individual media. Rapid developments are currently taking place in remedial methods, and therefore distinctions will be made between well-proven methods and new methods. Further assistance in selecting remedial techniques can be found in Appendix 9.1 which includes a table with costs, advantages, and disadvantages for each remedial measure, as well as the contamination components which can be treated with each method. The extent to which a given remedial measure can be applied at a specific location depends on a number of location-specific factors. Therefore, the choice of method should be closely linked to evaluation of operation and evaluation of the completed measures, cf. Chapter 10.

At the present time, rapid developments in remedial techniques are taking place. Information on alternative remediation technologies is therefore extensive. In addition, the Internet is a new source of knowledge which will be used increasingly in the future. Current addresses which are relevant, e.g. US-EPA (Hazard waste clean-up information) which have the following path http://clu-in.com (March 1997) or the Danish ‘grundvand på Internettet’: http://inet.uni-c.dk/-nnr-vm/ (March 1997), have many relevant ‘links’ to other addresses.

9.1 Clean-up objectives

If the risk assessment (see Chapter 5) concludes that there is a health risk in relation to land use, indoors or out, or that the groundwater or recipients are threatened, then remedial measures should be implemented.

9.1.1 Remedial measures in relation to land use

Remedial measures in relation to land use should aim at removing or cutting-off contamination, and preventing or reducing exposure.

This can be achieved through the following remedial strategies:

- Excavation of the contaminated soil with subsequent off-site (ex-site) or on-site treatment.
- In-situ treatment of soil and groundwater.
- Pumping groundwater near the surface.
- Construction measures for reducing indoor air exposure to volatile contaminants.
- Equipment to prevent or reduce exposure in the outdoor environment, and to prevent contamination from spreading /14/.

Furthermore, exposure can be reduced by changing or adapting land use to the actual conditions.

9.1.2 Remedial measures for groundwater and recipients

Remedial measures for the groundwater and recipients aim at reducing or preventing spread to the groundwater aquifer and recipients.

This can be achieved through the following remedial strategies:

- Excavation of the contaminated soil with subsequent off-site (ex-site) or on-site treatment.
- Active in-situ treatment of soil and groundwater.
- Pumping strategies with possible subsequent water treatment.
- Immobilisation of the contaminants (sealing, stabilisation, capping, cutting-off, fixing, vitrification).

9.2 Remedial measures for soil contamination

9.2.1 Overview of remedial methods

Developments in remedial methods are moving rapidly. There is a lack of documentation of the effects under Danish conditions for many of the new methods. This overview is therefore divided into well-tried methods and methods with potential application in Denmark.

There is a distinction between in-situ, on-site, and off-site (ex-site) methods. The following remedial methods have been started and completed for soil contamination in Denmark /31, 32/:

- Excavation and disposal of soil at central treatment facilities (ex-site).
- Excavation and soil disposal at landfills (ex-site).
- Excavation and soil disposal at treatment facilities (on-site).
- Soil vapour extraction (in situ).
- Forced leaching (in-situ).
- Methods using construction techniques and equipment (on-site, in-situ).

Furthermore, there are remedial techniques which have not been fully tested, but which have varying potential in Denmark. The following can be mentioned /31, 32/:

- Bioventilation
- Biological soil treatment (inoculation technique)
- Detergent leaching
- Immobilisation (vitrification, stabilisation)
- Electrokinetics
- Steam stripping
- Chemical treatment
- ‘Pneumatic fracturing’

Method selection depends on many factors, e.g. type of contamination, location of contamination, soil type, geology and hydrogeology, time available for clean-up, effect of clean-up and acceptable residual contamination, land use and lay out, working environment during remedial measures, costs of the methods, and last but not least, documentation of the methods application. Furthermore, the environmental effects should be assessed so that the best environment is achieved for the resources invested. It is assumed that all the remedial measures proposed eliminate the identified risks.

In addition to the descriptions below, the review of remedial methods has been collected and simplified in a chart in Appendix 9.1, where examples of costs for each method are also displayed.

9.2.2 Excavation

Excavation is by far the most common remedial method in the case of soil contamination. The contamination is removed, usually by an excavator, under controlled conditions, until the sides and bottom of the excavation are sufficiently clean. This is determined by the acceptance criteria for the specific contamination, which must be met at the completion of excavation work, cf. assessments described in Chapters 5 and 6.

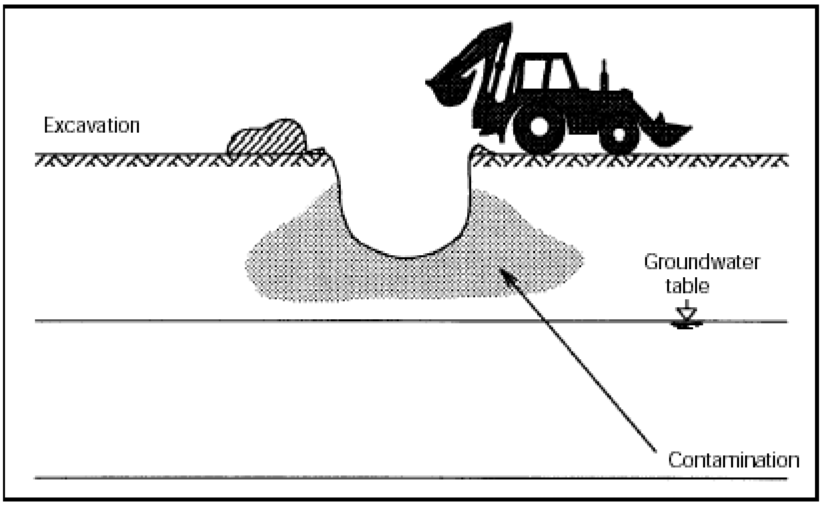


Figure 9.1
Excavation of soil contamination

In order to ensure the stability of buildings etc. the foundation norm (DS 415) must also be followed in all excavation work /33/. An example of how this is done in practice is described in Appendix 9.2.

The criteria for excavation must be documented using analyses of soil samples taken from the sides and bottom of the excavation. For excavation, the operation and evaluation phase occurs simultaneously with the remediation phase. The method is described in more detail in Chapter 10. In order to ensure that the requirements are complied with, the excavation must be supervised by environmental experts. This supervision is described in Section 8.5.

Until the early 1990’s, there was no real alternative to excavation which explains why the method is still the most widely used method today. The advantage of the method is that it is quick and well documented. More or less all completed remedial projects for soil contamination have been carried out with the help of excavation. Furthermore, excavation is applicable for all types of soil and contamination. The disadvantages of the method are the resulting environmental effects.

9.2.3 Treatment methods for excavated soil

Since 1990, local authorities have been responsible for assigning the disposal of all industrial and commercial waste, including contaminated soil and contaminated construction waste (oil separators, tanks, foundations etc.). With regard to disposal, soil is often subdivided into classes of contamination /34/.

There are several central soil treatment facilities, most of which treat soil contaminated with organics by microbiological degradation (stack composting or land farming). If there is a significant heavy metal, cyanide, or tar content, thermal treatment for, e.g. tar products, is possible at some treatment facilities or at various power stations. Chemical extraction (carried out by a single firm in 1997) treats soil for tar, pesticides, cyanides, and heavy metals in addition to lighter contamination. Finally, there is the possibility of landfilling heavy metals, for example. Stripping of highly volatile contamination is only applied to a very limited extent.

Contaminated soil can be landfilled. Landfilling can, however, only take place if it is authorised by both supervision authorities at the landfill site, and the local authority from where the contaminated soil originated.

Some soil treatment firms offer to treat excavated soil on-site using the same principles as central plants (land farming and, to a lesser extent, stack composting). Furthermore, on-site mobile treatment equipment (thermal methods, stripping, soil rinsing, etc.) has been tested to a very limited extent /31/. Finally, forced leaching has been tested at a single site (water is percolated through the contaminated soil, enclosed in membranes. The water is cleaned and recirculated) /31/.

Experience of methods in Denmark is not exhaustive. Completed projects indicate that methods require large amounts of soil before they become attractive, and they require generally sandy soil and lighter organic contamination. The disadvantages of the methods include space requirements, expensive construction requirements (e.g. leachate collection systems), extensive time requirements, whether excavations may remain unfilled for longer periods, and odour and noise problems.

9.2.4 Soil vapour extraction

Soil vapour extraction is primarily a physical removal (stripping) of highly-volatile xenobiotic organic substances from the unsaturated zone using a vacuum. This method is the most frequently applied in-situ method in Denmark.

A number of active ventilation screens are installed in the unsaturated zone, and they are subjected to a vacuum using a ventilator. Highly volatile substances are in this way sucked out of the soil. In some cases, passive screens can be more appropriate in order to control the air flow (particularly for contamination under buildings).

Depending on the type and concentration of contamination, it is often necessary to clean the extracted air, usually using carbon filters. If there is benzene at the site, such cleaning should usually be performed because of the low air-quality criteria for this substance.

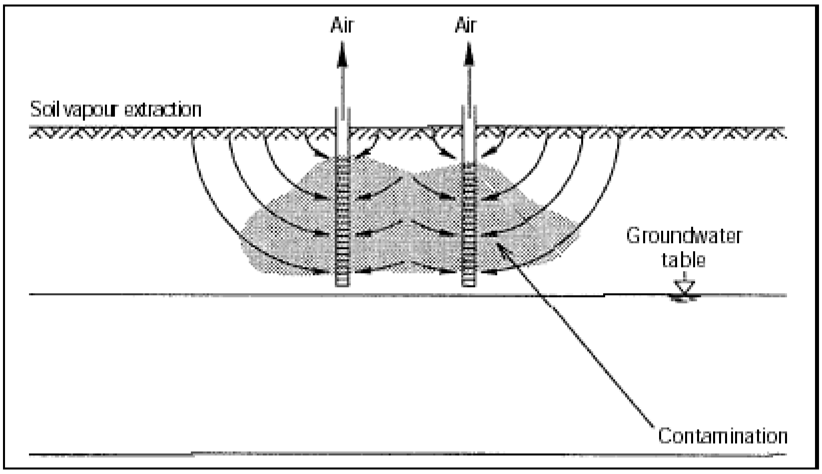


Figure 9.2
Soil vapour extraction

Soil vapour extraction is most appropriate for remediation of highly volatile organic contamination in looser soil types. In order to design soil vapour extraction properly, air-permeability tests should always be performed /35/, and bio-activity tests can also be of assistance. Bio-activity tests can document the potential for biodegradation as well as assist in determining the distance between the extraction drillings (radius of influence). The location of the contamination can be decisive in choosing this method, as this method is well suited to contamination which lies close to, or under, buildings. The time required to remediate can vary depending on the soil conditions, the type of contamination, and the stop criteria (typically from five months to several years). Many remediations of this type have been completed and approved in Denmark (1997). This method requires that stop criteria and evaluation methods are set from the outset, cf. Chapter 10.

9.2.5 Treatment methods for extracted air

Normally, the extracted air must be treated before emission. Permitted emission levels appear in ‘Orienteering’ No. 15, 1996, from the Danish EPA /11/. The following treatment methods are typically used for extracted air:

- Granular active carbon (GAC)
- Catalytic oxidation
- Direct incineration
- Biological filters

It is most common to use activated carbon filters to treat the extracted air before it is emitted. The advantages of a activated carbon filters are that the method is easy and safe, both for small air flows and high concentrations. Furthermore, purchase of the equipment is relatively inexpensive. On the other hand, the equipment is expensive to operate, and there are often noise problems. Activated carbon filter equipment also requires a lot of supervision, particularly at the start where carbon must be changed frequently. Finally, it is important to note that the effect depends on temperature and the composition of the contamination.

Catalytic oxidation is cheap to operate as the method is self-controlled and removal takes place with no critical by-products. On the other hand, purchase of the equipment is expensive and the method requires high concentrations of contaminants.

As with the catalytic method, purchase of biological filter equipment is expensive, but operation is inexpensive. On the other hand, this method is relatively sensitive in operation. Direct incineration is also expensive.

9.2.6 Bioventilation

Bioventilation is the aerobic microbial degradation of xenobiotic organic substances in the unsaturated zone, for example through the addition of atmospheric air or oxygen. A number of bioventilation screens are installed in the unsaturated zone. Air is blown in using a ventilator, and decomposition of the contamination is stimulated. Usually, a number of passive ‘air-emission screens’ are located at appropriate distances depending on the characteristics of the contamination. Bioventilation stimulates biodegradation by blowing in air, unlike soil vapour extraction where contamination components are sucked out of the soil.

Projects regarding bioventilation have been approved and are in operation in Denmark, but as yet, there is no knowledge of completed remediation in Denmark (1997). The USA has performed many remediations using this method. It seems that the method is best suited to remediation of lighter, aerobically degradable organic contaminants (mineral-oil products and solvents, but not chlorinated solvents) in permeable soil types. The method is also most suitable for substances with a low to moderate vapour pressure. Otherwise, there is a risk that the substance will be stripped before it is degraded. Air permeability tests and bioactivity tests should be performed when designing the equipment, with a view to ascertaining the air flow and the degradation potential of the site /35, 36/.

The location of the contamination is significant. For example, this method should be considered where contamination is located under or close to a building.

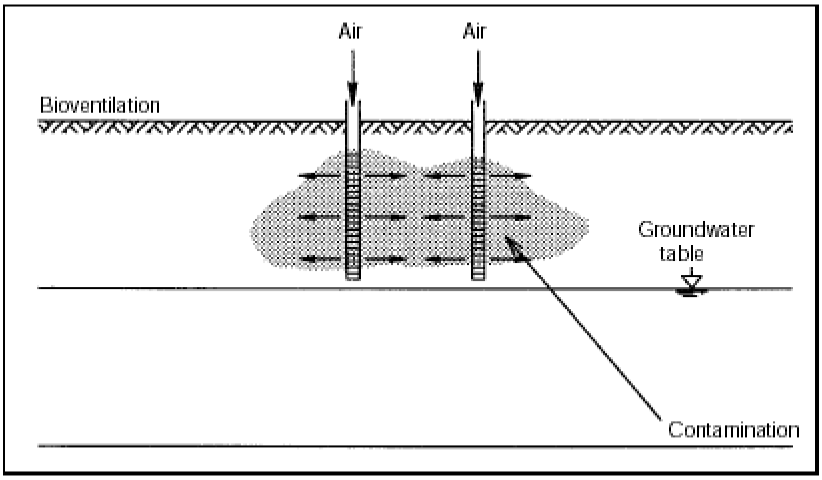


Figure 9.3
Bioventilation

This method is not particularly well documented in Denmark, and the time required for clean-up is still unknown. However, there is no doubt that the method is beneficial as a supplement to, or in combination with, other methods such as soil vapour extraction or groundwater pumping.

9.2.7 Forced leaching

Contaminants are forced to leach by artificially increasing the infiltration of water through the contaminated area, possibly by recirculating abstracted water. It may be beneficial to add nutrients, bacteria, and oxidants to the water to stimulate degradation, or detergents may be added to increase bioaccessibility (detergent leaching).

The leach water is infiltrated, either via leach fields, via sprinklers, or directly into the saturated zone. The leach water will usually be treated water abstracted from the contaminated zone, or uncontaminated water abstracted nearby for the purpose of hydraulic control.

This method works best in combination with other methods, normally remedial pumping. In this case, the abstracted water can be used for leaching after treatment, and hydraulic control is ensured. The method appears to be appropriate for remediation of soluble and bio-degradable contaminants in relatively homogenous, sandy deposits with well-defined hydraulic conditions.

A few projects using this method have been completed /31, 32/. However, detergent leaching is still under trial. It would seem that the mobile components in the contamination are removed relatively rapidly (within months), but total remediation seems to be impossible in practice if this method is used alone. Furthermore, it should be noted that there can be operational problems with this method, and in many cases these have resulted in problems as a consequence of clogged screens, etc. due to precipitation of iron and biological growth. Substances which are added to the infiltration water can give pollution problems, e.g. bacteria, detergents, etc.

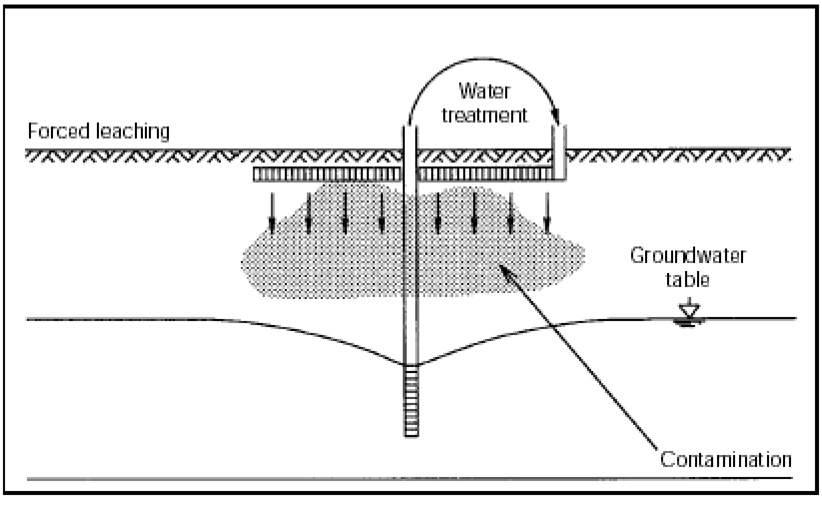


Figure 9.4
Forced leaching

9.2.8 Immobilisation

Instead of removing contamination, it is possible to hold the contamination in place so that the area can be used for specific purposes. In Denmark, constructional methods are often used, e.g. asphalt and paving which prevent surface contact with the soil ensure that the further downward movement of the contamination is minimised.

The contamination may be sealed in by the placement of membranes made of synthetic materials or very low-permeable materials (clay or bentonite). In especially complicated cases of contamination, vertical barriers may be applied to prevent horizontal spreading of the contamination, for example using membranes (in open excavations), sheet piling or vertical barriers of bentonite/concrete bentonite/soil, and slurry walls or drilling techniques (grouting and Deep Soil Mixing). There are therefore many well-tested techniques to contain the contamination using impermeable barriers, as they are based on geotechnical techniques often applied abroad.

Remediation has been executed abroad in a small number of situations using in-situ vitrification, where the soil is heated by an electric current and converted to a glass-like mass. In addition, in-situ stabilisation is often applied in the USA, but has only been tested in a limited number of situations in Denmark /31/. Attempts have been made to reduce leaching by mixing a stabilising agent with the soil (e.g. bentonite, cement, or lime).

For sealing, it is crucial to ensure that the material is impermeable. However, groundwater can still be affected by the contamination as horizontal movement of groundwater in the water-bearing layer occurs independently of rainfall at the site. In addition, diffusion of volatile substances must not be ignored. Therefore, the method is not applicable as the only solution for highly volatile contamination. Emplacement of membranes should be supplemented with systems for collecting and draining precipitation.

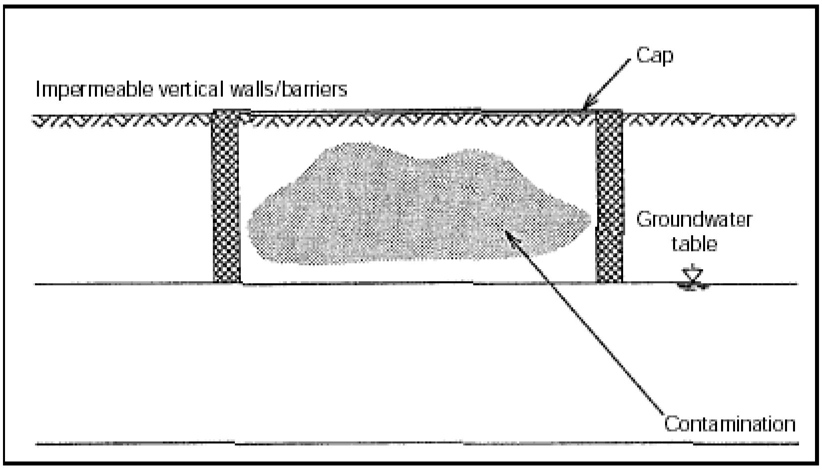


Figure 9.5
Sealing contamination

9.2.9 Bioremediation

In bioremediation, optimal conditions for degrading contamination are created in the soil. This can be done by adding appropriate micro-organisms (inoculation technique) or by improving living conditions for naturally-occurring bacteria (stimulation technique), for example by adding oxygen or detergents (increases biological availability by increasing solubility).

Remedial projects have been started which use the stimulation technique (bioventing, see above), while the inoculation technique is still at the experimental stage in Denmark. In principle, most organic substances can be degraded by micro-organisms, except substances such as PCB, chlorinated dioxins, heavy metals, and high-molecular PAHs. Certain conditions regarding the physical-chemical relationships in the soil matrix must also be fulfilled. These include oxygen content, inorganic nutrient content (e.g. ammonium and phosphate), availability and toxicity of the xenobiotic substances, temperature, and pH. Additionally, water content and soil type are also significant (soil with a high clay content is not appropriate). Tests have generally resulted in heterogeneous and high-concentration residues, and there are also problems with the long duration of the remediation process. This is why biological in-situ inoculation methods are not yet commercially utilised /37, 38/.

9.2.10 Other in-situ methods of soil remediation

Electrokinetic soil remediation removes heavy metals or organic contamination from soil. The technique forces the heavy metals contaminants out of the soil by electrokinetic processes which arise when the soil is subjected to an electric field (electromigration). Organic contamination can also be removed by electro-osmosis. Pilot tests are currently in progress in Denmark /39/, and the method may have a commercial future for removing heavy metals. Full-scale execution of the methods below has only taken place a very limited number of times abroad and therefore they are not immediately commercially accessible in Denmark.

In steam stripping, the soil is loosened using two counter-rotating drill bits. Steam and compressed air are pumped down through the bits and into the soil. In this way, volatile components are stripped from the soil and evaporate to the surface. The method has not been attractive under Danish conditions, primarily because it is an expensive method which demands a lot of energy. Furthermore, the method requires that objects larger than about 0.3 m are removed from the soil, and that the site does not slope by more than 1 per cent.

Through the infiltration of active substances, contaminated soil is degraded into less toxic substances. This method has not been applied in Denmark as there have not been any pilot-scale trials. The method requires soil with a high permeability. Furthermore, there will probably be problems in cleaning sufficiently to meet specific quality criteria.

The soil is exposed to compressed air via soil borings with a view to increasing the permeability of the soil. The method can be combined with other techniques, e.g. venting. This method has not been utilised in Denmark, primarily because there is no documentation of the applicability of the method.

9.2.11 Running-in soil remediation installations

If soil remediation uses technical installations, this equipment must be run-in once it has been established. For example, in order to optimise air flows, running-in of venting installations typically includes recording amounts of air pumped and pressure measurements in monitor wells. At the same time, a certain number of chemical analyses are taken. Running-in should be concluded with the completion of an operating manual.

Operation includes maintenance of the technical installations, monitoring the amount of contamination, and regular assessment of the equipment and contamination collection. In order to ensure optimal operation, an equipment register, a calendar of activities, descriptions of work, and regular status reports are prepared. Operation and evaluation for individual techniques is described in more detail in Chapter 10.

9.3 Remedial methods for groundwater contamination

9.3.1 Overview of remedial methods

There are various principles and methods for remediation of groundwater contamination. The choice of remedial strategy in a specific situation depends on the following factors:

- Type of contamination (phase distribution, density, etc.) and composition.
- Position of the contamination (horizontal and vertical), as well as the contamination's extent and severity.
- Hydrogeological conditions (hydraulic parameters, type of aquifer, hydrological basin, etc.
- Time required for remediation.
- Site conditions.
- Necessity of hydraulic control.
- Amount of investment and operating and maintenance costs.

The most common remedial methods known today are:

- Ordinary pump-and-treat from screened wells.
- Separation pumping from specific levels.
- Pumping with multiple pumps in several phases.
- Skimming LNAPL contamination from screened wells.
- Pumping from drainage systems.
- Pumping from suction-probe equipment (including 'bioslurping').
- In-situ methods (including air sparging, adding oxidising agents, reactive walls, vertical barriers).

9.3.2 Pump-and-treat

Pumping from deeper aquifers is typically performed from screened wells.

In order to bring a contamination under hydraulic control, a pumping strategy must be prepared. A pumping strategy includes the following /40/:

- Location of pump wells
- Number of pump wells
- Pump yield
- Pump levels

Depending on the situation, a number of different methods are available to fulfil the pumping strategy. These include ordinary pump-and-treat from screened wells, separation pumping, skimming, injection, recirculation, or possibly a combination of methods.

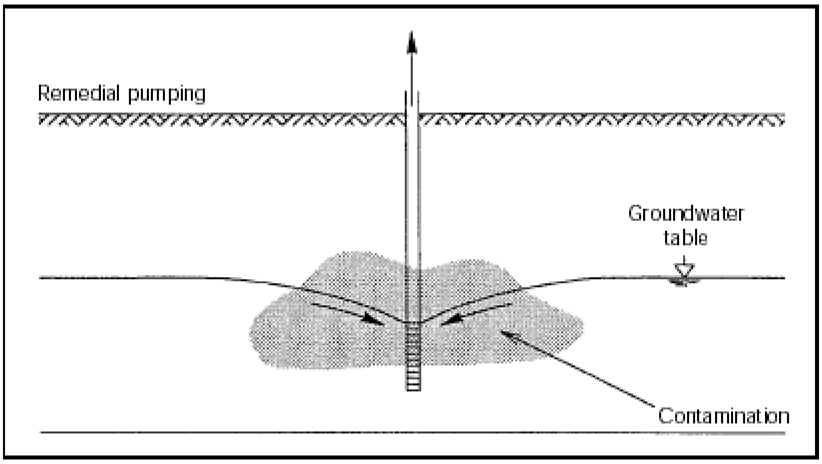


Figure 9.6
Remedial pumping

In cases where contamination consists of a light non-aqueous phase liquid (LNAPL) on the groundwater, it is usual to remove the LNAPL by skimming before alternative remedial methods are started. If a LNAPL e.g. petrol and oil is present, extensive drawdown of the groundwater table should be avoided as this will cause the contamination to smear the exposed soil where it cannot be removed using simple methods. Using several wells with a smaller drawdown, possibly with the aid of vacuum to remove air and water simultaneously, can be the optimal solution in these cases.

In cases where there is groundwater contamination near the ground surface, it is often advantageous to utilise drains connected to a collection sump from which groundwater is pumped. This solution is particularly relevant in connection with excavation, as the method usually requires extensive excavation. Drilling horizontal drains can be a solution in some cases.

Suction-probe equipment may be appropriate for short-duration pumping in sand aquifers near the surface (max. 5-7m delivery head).

Bioslurping is a relatively new method, which in principle is a further development of the suction-probe technique. By using a vacuum, both liquid and air are removed at the same time through an adjustable suction pipe which can be located in conventional wells. The well opening must be sealed to maintain a vacuum.

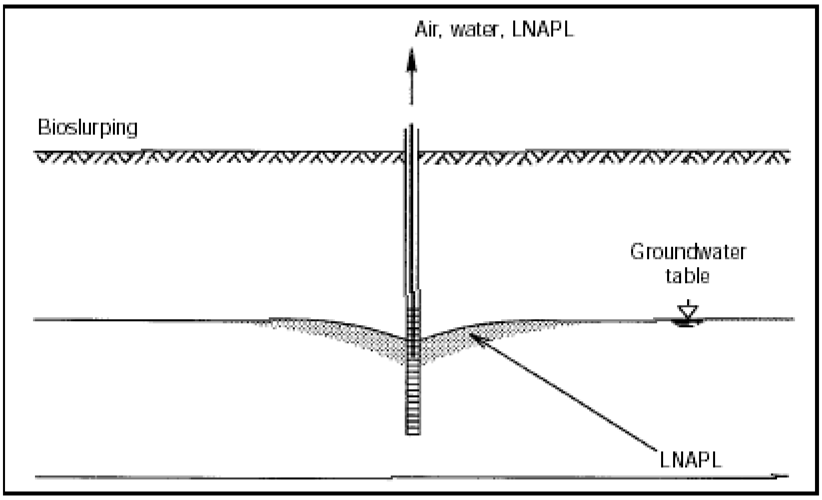


Figure 9.7
Bioslurping

There are various methods which can be used to optimise pumping strategies. The strategy is usually set on the basis of the location of water abstraction wells and their capture zones. The overall groundwater flow direction is normally determined through measuring the potentiometric surface. Through pump tests, the hydraulic parameters of the aquifer, transmissivity, specific yield, and leakage can be determined. Furthermore, the vertical variations in the reservoir can be determined by geophysical logging. By performing conductivity and temperature logging, variations in ion distribution and temperature can be determined, and flow logs determine variations in inflow. In addition, there are a number of logs which provide various geological information on the formation (gamma, electricity, resistivity, and conductivity logs).

With other hydrogeological data and knowledge of the extent and nature of the contamination, these data can be used to determine the optimal pumping strategy. Groundwater flow and contaminant transport models can be used where conditions require information on alternative strategies. There are examples of several types of numerical model which have different applications. Both two and three-dimensional models are available. Today, three-dimensional models are usually used, which can perform both stationary and dynamic simulations of the scenarios desired.

Pump types and technical equipment to control pumping depend on the situation. There are numerous types of pump. Submersible pumps are often used in deeper aquifers, while vacuum pumps and centrifugal pumps are suitable for aquifers close to the surface (up to a depth of about 7m). There are various technical accessories to ensure hydraulic control by maintaining the required groundwater potentials, e.g. level controls, pressure transducers, timers, or electrodes.

Vast experience with the pump-and-treat method has been gained. Many of these cases have now been completed. However, many cases have demonstrated complications in achieving stop criteria, and total remediation up to groundwater criteria is not a realistic goal. On the other hand, using hydraulic control, it is possible to prevent groundwater contamination from spreading towards abstraction wells etc., and at all events, a large amount of the contamination is removed.

Following construction, a running-in phase (implementation) for the remedial installation begins. This phase is aimed at optimising operation. Instructions for running-in the technical equipment are prepared (includes recording electricity consumption, checking pumps, documenting pump yield, documentation of water treatment, etc.). In addition, instructions for contamination removal are prepared (includes recording pump performance, recording water/air quantities, taking water table measurements, analysis programme, and results).

When the running-in phase has been completed, compliance with success criteria is assessed (water table levels, contamination concentrations, etc.). Any necessary changes are then made (technical equipment, pumping strategies, groundwater models, etc.). Operation and evaluation of the pump-and-treat system are described in Chapter 10.

9.3.3 Discharge and treatment of the abstracted groundwater

In some cases, contaminated water can be discharged to the nearest sewage treatment plant or to less-sensitive surface waters. In other cases, water treatment is required before discharge. This requires individual assessment of the situation in question. For discharge directly into sewers, analyses should be carried out to prevent possible damage to the treatment plant. In addition to the type and concentration of contamination, water quantities and content of organic matter are crucial factors in determining whether discharge into the sewage system is possible. At the same time, assessments must be made as to whether discharge can present problems for the work environment of sewage workers.

Prior to discharge to sewage treatment plants, authorisation must be obtained from the supervisory authority for the sewage treatment plant in question. The supervisory authority will determine whether the plant is able to accept the water. If there are problems in accepting the amounts of water, it may be possible to discharge the water at night when the other demands on the plant are usually low. Significant sewage expenditure must be expected for discharge into sewers. Expenses can vary from local authority to local authority.

Treatment may be required before discharging contaminated groundwater to sewers, a surface water recipient, or a reservoir. The method of water treatment used depends on the contaminant in question as well as water quantities and quality requirements. Therefore, these methods must be assessed individually for each specific case. There is a distinction between on-site treatment, where the abstracted groundwater is treated using equipment placed on the contaminated site, and in-situ treatment where the groundwater is treated in the aquifer without being pumped. The following on-site treatment principles are generally recognised and accepted for remediation of contaminated groundwater. For special problems, please refer to the Soil Project on land use and remediation of contaminated groundwater /41/. Appendix 9.1 provides examples of costs of the different water treatment methods.

Gravimetric separation of contamination from pumped groundwater has long been applied in oil/petrol contamination. Traditional oil-water separators, however, often have limited effectiveness in that residual concentrations can be as high as 100 mg of oil product per litre at the outlet /41/. A further development is the coalescence separator where a built-in synthetic material gathers the small drops of oil into larger drops which rise up to the surface of the water. In this way, removal effectiveness is significantly improved with residual concentrations as low as 20 mg of oil product per litre.

If the pumped water contains stable emulsions and high concentrations of dissolved components, equipment for destroying the emulsion can be used. Chemicals are added and the effect which is achieved, depends on dosage and residence time. This system requires a greater degree of on-going evaluation during operation. The reader is referred to more detailed rules regarding the type and functional requirements for discharge systems with separators in DS 432 “Norm for discharge installations” /42/.

Treatment of water by using filters is a prevalent and well-documented method. The choice of filter material depends on the specific contamination situation. Traditional sand filters are widely used as pre-treatment for the removal of iron, manganese and ammonium prior to special processes, but are also used for the biological degradation of organic contamination. Pre-treatment for iron and manganese removal should always be considered in water treatment as this is often a prerequisite for the subsequent special processes can be carried out as intended.

Membrane filtering is carried out by pressing water through a semi-permeable membrane which retains molecules larger than the water molecule. This method is used for removal of salts and heavy metals /41/. Adsorption filtration with activated carbon as the adsorbent is a widespread method for removal of all types of organic contaminants.

All of these filter technologies have a common need for backwashing and/or cleaning of the filter material and occasional replacement. The effect of the filter is reduced with time as the filter material slowly looses its ability to absorb/adsorb the contaminants or is clogged. Therefore, considerable operational expenses should be expected in connection with filter cleaning, backwashing and/or replacement.

For volatile organic contaminants, a process where contaminants are transferred from the water to the air of the stripping system may be used. The effectiveness of this process depends primarily on the design of the system and the contaminant’s vapour pressure and water solubility /41/.

In photochemical oxidation, hydrogen peroxide and/or ozone is activated by ultraviolet radiation. In this manner, substances that provide a strongly oxidising environment and destroy the contaminants are formed. The products of degradation are water, carbon dioxide and volatile easily degradable organic acids. The method is useful for destroying most types of organic contaminants, such as benzene, mineral oil products, solvents, pesticides and cyanides /41/. The effectiveness depends on the residence time of the contaminant in front of the UV light. This residence time can be varied, depending on the discharge requirements.

Biological treatment of oil products is well-known from sewage treatment plants. In Denmark, there are only a few good experiences with on-site biological treatment of contaminated groundwater. Variations in the concentration level and temperature, degradation rates and stability are all factors which make the operation of small treatment units very difficult to optimise.

9.3.4 In-situ remediation methods for contaminated groundwater

Air sparging has recently begun to be used in Denmark. Air sparging implies physical removal and microbial degradation of contamination in the groundwater by blowing, for example, atmospheric air below the groundwater table. Air is blown below the groundwater table so that volatile components are stripped and transferred from the water phase to the unsaturated zone, where they must be removed using other techniques. Furthermore, microbial decomposition in the groundwater zone is stimulated because of the added oxygen.

Only a few completed air-sparging remediations are known (1997), but the method is considered to have a future in Denmark if it is combined with other methods, e.g. soil vapour extraction for organic volatile contamination under homogenous geological conditions. The geology is a decisive factor in that a reasonable amount of homogeneity in the media is required. This is particularly important for remediating chlorinated solvents since stripping is the only removal mechanism.

In order to determine whether a site is appropriate for this method, and to design the system, a well-designed pilot test should be conducted in the form of air-sparging/tracer tests in the reservoir where the system is to be installed /32, 43/.

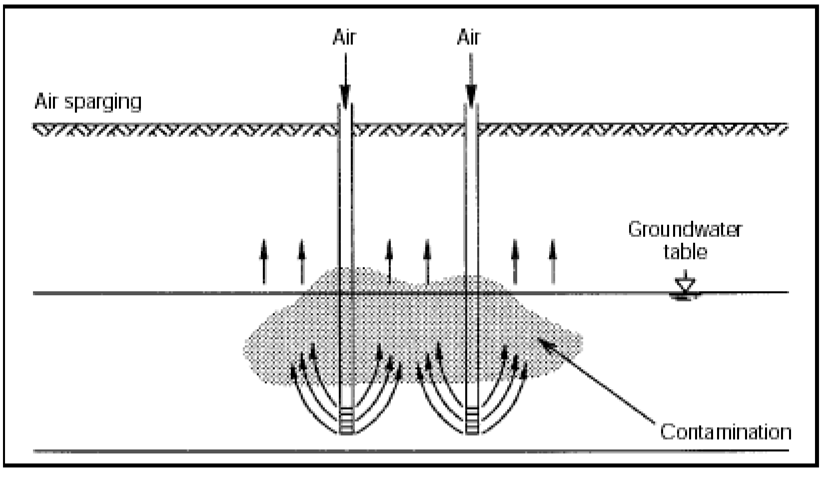


Figure 9.8
Air Sparging

A related method, developed from air sparging, is biosparging. In this method, the primary objective is to stimulate the biological process. Here, the oxidising agent is added in pulses under lower pressures.

A new method, which was first used in remediations in Denmark in 1997, is to add the oxidising agent or Oxygen Release Compound (ORC) to the groundwater zone. The method is relatively new in the USA, but it has become very successful over a short period. The method is inexpensive and environmentally friendly, and it will probably become widespread in Denmark.

It is possible to cut-off groundwater contamination by establishing vertical barriers in the groundwater aquifer. This can be done using various methods such as sheet piling, excavation methods, slurry walls, drilling methods, deep soil mixing (DSM) and grouting. The different methods use different materials for the barriers such as betonite, and possibly in combination with different types of plastic panels (geo-membranes). The methods have been applied at many sites abroad, but have yet to be used in Denmark. It is necessary to be aware of the physics and location of the contamination, as well as possible problems with groundwater backing up. Therefore, it can be beneficial to construct barriers in the form of a funnel leading to a permeable gate (funnel and gate technique) where a reactive permeable barrier can be constructed in the gate area /44/.

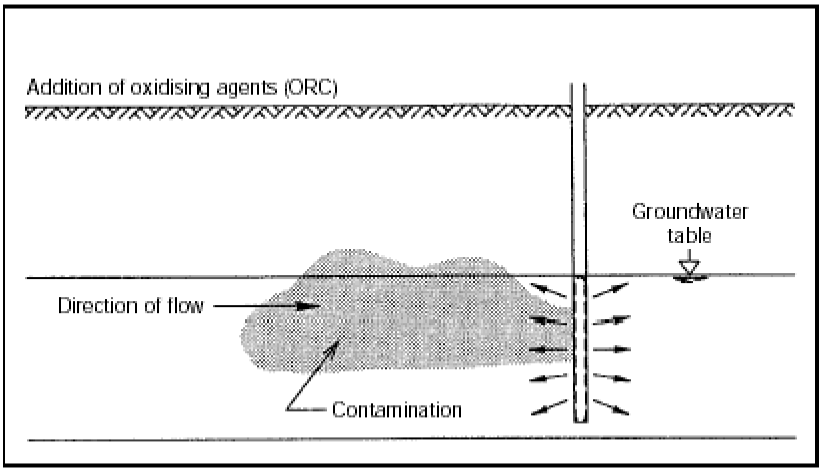


Figure 9.9
Addition of oxidising agents

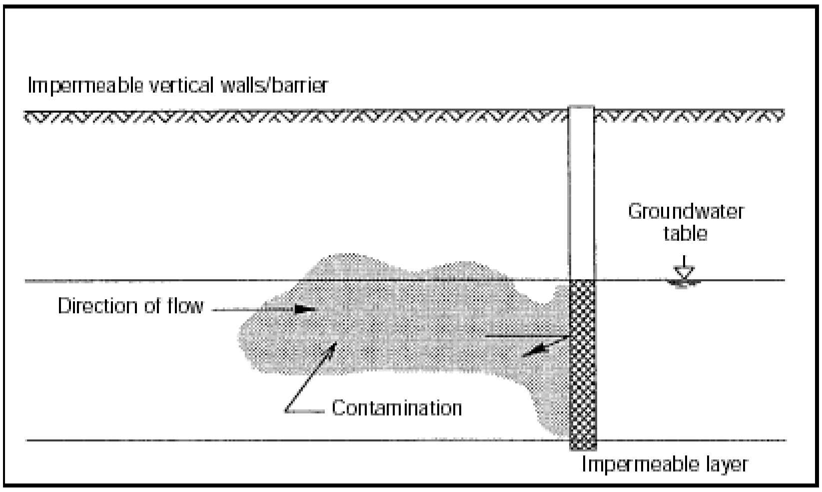


Figure 9.10
Impermeable cut-off walls

Reactive permeable walls are barriers which allow the passage of groundwater, but which degrade or remove contamination from groundwater during the passage. The method is at an experimental stage in Denmark, but it is used in the field in the USA for degrading chlorinated compounds with iron filings as the reactive material in the barrier. In addition, materials can be used with particularly high sorption properties, e.g. clay minerals or active carbon. The barriers may be disposable or reusable modules. This method may have a future in Denmark.

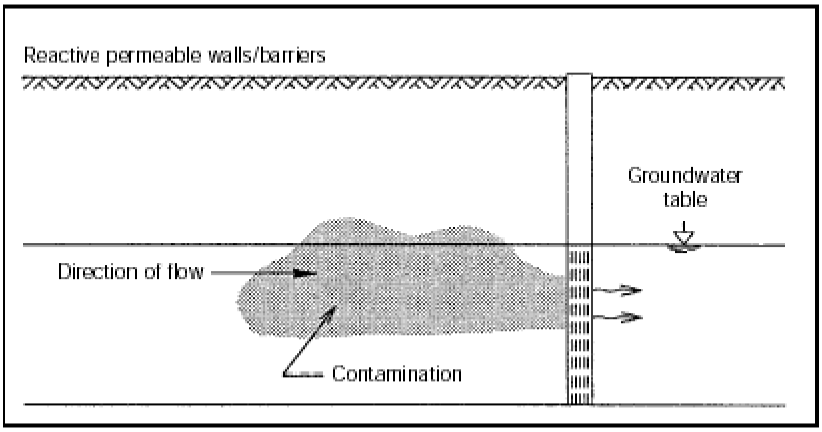


Figure 9.11
Reactive permeable walls

9.4 Remedial methods for contaminated soil gas

9.4.1 Buildings on contaminated areas

As far as possible, construction of new buildings should be should be planned so that they are not located on contaminated areas, as this can create obstacles to a subsequent remediation. Land use should be planned so that contaminated areas are not used for very sensitive land uses. The environmental authorities can, however, allow contaminated sites to be used for specific purposes without requiring total remediation. All new construction, including construction on contaminated sites, must be conducted according to the guidelines on radon in new construction from the National Housing and Building Agency /14/.

In existing buildings with unsatisfactory concentrations of contaminants with respect to health which stem from contaminated soil, a practical solution is often improved ventilation in crawl spaces, the gravel drainage layer under floors or in cellars, although it should be noted that strong ventilation in cellars may increase convection of volatile xenobiotic substances through cracks and leaks.

In outdoor areas which are frequented by people, there should be protection from exposure to contamination, primarily from dust from contamination near the surface, and more rarely from evaporation of volatile contamination. In many cases, the area can be sealed with a cover such as asphalt or possibly by replacing the upper layer of soil, depending on the situation.

9.4.2 Construction technology measures

In order to prevent volatile contamination from being transported up in buildings, there are various options for ventilating gravel drainage layers under floors, cellars, or basements. Moreover, ventilation generally can be improved in rooms. For ventilation drains, drainpipes are laid out in, for example, the gravel capillary-breaking layer under floors. Ventilation pipes are connected to a collection well where a ventilator may be installed, if this later proves necessary. Ventilation pipes may also provide passive ventilation and should therefore be extended over the ground surface with possibilities for venting into the free air. In planning new buildings, natural air renewal can be used by constructing crawl spaces or elevating the buildings.

Ventilation is a very widespread method, and is usually adequate for most contamination types. Ventilation solutions are easy to include in construction projects and are also inexpensive to install. Experience also shows that it is often not necessary to commence active ventilation since passive ventilation is usually sufficient.

As a supplement or possible alternative to ventilation solutions, diffusion-inhibiting synthetic membranes may be laid out. There are different products which may be used, depending on the contamination. It must be ensured that the membrane is appropriately welded, and that it stops the contamination in question.

Transport of contamination into the building can also be prevented if the choice of materials includes reinforced-concrete foundations and floors of at least ‘moderate environmental quality’.

9.5 Remedial methods for landfill gas

If there is a health/safety risk for occupants in a building on or near a landfill, remedial measures should be carried out so that landfill gas is prevented from reaching the buildings, or so that concentrations do not exceed acceptable levels and/or so that an early warning is given by gas alarms.

The objectives of remedial measures for landfill gas are to ensure that methane does not present risks of explosion, and that carbon dioxide does not present a toxicological risk. This is achieved by checking gas flows and thereby ensuring that the gas does not flow towards the buildings, but is led away from the critical area.

As with all other contamination, new construction should be located outside the risk area, if possible. For more detailed considerations, refer to Report No. 69 on landfill gas from the Danish Environmental Protection Agency /17/.

Remedial measures can include various principles. The following methods are used to prevent the flow of gas towards buildings.

- Gas barriers (constructed between the source of gas and the building, and normally comprising a gas-tight cut-off membrane placed in a ditch on the side of the ditch that is closest to the building. Synthetic and natural membranes are used).
- Permeable ditches (established between the source of gas and the building, and normally comprising a cut-off ditch with coarse material and possibly a gas-tight membrane in the side of the ditch closest to the building. The gas is vented either passively or actively (in drains)).
- Venting drains (the gas is vented between the building and the source of gas either passively or actively).
- Venting wells (the gas is vented between the building and the source of gas either passively or actively).

The following methods are used to prevent gas from entering into buildings.

- Sealing buildings (the building is sealed against gas convection, for example by using membranes and sealing cracks in concrete).

- Changing the pressure gradients (e.g. buildings can be slightly pressurised above atmospheric pressure). This method is not recommended, however, for buildings which are damp.
- Ventilated drains (drains under the building are ventilated on the same principle as used for preventing other volatile contamination described in Section 9.4.2, or ventilation is established around the buildings and in dead space and sewers. The system can work either actively or passively).

Furthermore, as an extra safety measure in risk areas, systems should be established for monitoring the concentration of gas, and gas alarms should be installed to ensure the effectiveness of remedial measures, as well as to prevent possible injury to people. The monitoring system could start ventilation if high gas levels are measured.

The choice of method depends on the specific situation, and usually a combination of several methods is possible. The location of the contamination in relation to the building is crucial to the choice of method. If the building is located outside the landfill, the problem can often be solved by preventing gas migration to the building, possibly using drains. If the building is located directly on the landfill, construction methods must be used (to prevent entry of gas into the building).

There are both active (ventilators) and passive (natural ventilation) systems. The main rule is that ditches usually function passively, while wells must usually have active ventilation installed to ensure that an adequate radius is affected.

9.6 Residual contamination under houses

If there are strong reasons for leaving residual contamination because it is impossible to remove, a risk assessment of the residual contamination should be performed before decisions are taken, cf. Section 5.2. This should always be followed by administrative regulation so that inappropriate spreading is avoided.

In a project on residual contamination under houses /45/ some suggestions are provided on how contamination under houses can be managed, including which remedial measures are appropriate.

10 Operation and evaluation

10.1 Introduction and objectives

The nature of the evaluation phase depends on the type of remedial measures. With excavation, the evaluation phase is short term and takes place more or less simultaneously with the remediation phase. With in-situ remediation of soil, air or groundwater, the operation and evaluation phase can be longer term. Therefore, the evaluation it is often divided into an ongoing monitoring/operation phase, followed by an evaluation of compliance at the conclusion of the remediation. There is often a difference between the effect during operation and the lasting effect of remedial measures, and in these cases there must be a sharp distinction between evaluation during operation and evaluation of the lasting effects. Under circumstances where there is no immediate need for remedial measures, simple monitoring may be necessary. This applies in particular to groundwater aquifers. Evaluation measures are closely related to the individual media and remedial methods, and therefore the descriptions below are very method-specific.

The objectives of the evaluation phase is to evaluate and document the effect of completed remedial measures, cf. the guidelines presented in the planning phase. The objective of monitoring in cases where no remedial measures are carried out is to check that an unacceptable spreading of contaminants does not take place.

Stop criteria and monitoring programmes must always be set before starting a remedial measure. Stop criteria are based on the risk assessment, and are usually set during preparation of the outline project. When there is an operation phase, a monitoring programme must be prepared along with the outline project. During running-in of the technical equipment, an operating manual should also be prepared which contains all necessary information about the system, a description of the equipment, activities, and descriptions of work /40/.

10.2 Evaluating excavation

Excavation removes soil contamination either partly or completely. Evaluations are carried out simultaneously with the excavation in order to ensure that the soil is sorted into contaminated soil and clean soil (possibly in several classes of contamination). Evaluation should also ensure that residual contamination is in compliance with the excavation criteria (e.g. that the contamination level in the sides and bottom of the excavation are sufficiently low). Guidelines for managing the contaminated soil should be clearly stated in the project description, and before commencing work, they should be designed as instructions/action plans for personnel directly involved.

In order to ensure that requirements are complied with, excavation must take place under environmental supervision. The tasks of the environmental supervision personnel are described in Section 8.5.

There are three types of evaluations for excavation.

- Evaluation of the excavated soil.
- Evaluation and documentation of residual contamination after excavation.
- Evaluation of remediated soil.

10.2.1 Evaluating excavated soil

In order to optimise remediation of contamination, a clear excavation strategy should be agreed upon. The excavation strategy primarily depends on the results of contamination mapping in the investigation phase, in particular the homogeneity of the contamination. A strategy for sampling must always be established, outlining where samples are to be collected from, how often they are to be taken, and how samples are to be taken for field measurements and laboratory analyses.

Furthermore, attention should be directed to how the excavation is to be physically carried out. This is important with regard to geotechnical considerations, and is also often important with regard to evaluating the excavation. For example, it may be appropriate to describe how large an area should be removed at each level, the depth of soil removal at each level, the maximal extent of the excavation, what size the bucket on the excavator should have, etc.

The method and price of soil treatment depend on the type of contamination and its concentration. It is therefore necessary to document the contamination in the excavated soil by taking soil samples for analysis. The number of samples which should be taken for laboratory analysis depends on several factors, including:

- How homogeneously the contamination is distributed.
- The type of contamination (can the contamination be detected using simple methods, for example field measurements or visually).
- Whether the contamination is to be divided into several classes of contamination for different methods of disposal.
- How the soil is to be disposed of (soil for reuse or landfilling may require more analyses than soil for treatment).
- The total amount of contamination (small amounts require relatively more analyses than large amounts).
- How many investigations have been carried out before excavation started (for example, has the contamination been well-defined?).

In the design phase, a plan for the collection of samples is established. The plan may include the number of samples and the systematic pattern of distribution to be used in the area in question and divided into each layer of soil to be removed. In addition, the plan can describe collection of samples from an interim soil storage location, receiver facilities, or directly from the excavator bucket.

Requirements can either be defined as a number of samples per unit of weight, or per volume of excavated soil. In general, a conversion factor of 1.8 tonnes/m^3 can be used.

Soil treatment prices depend on the degree of contamination. In cases of very heterogeneous contamination, it can often be financially advantageous to collect many samples so that the soil can be sorted into contamination categories. Financial optimisation must provide the basis for how many samples should be taken in these cases. The supervision inspection journal and plan should always include details of where in the excavation a soil sample originates.

When taking samples with volatile substances, especially in interim soil storage location, it is necessary to be aware of losses of contaminants. Therefore, appropriate sample containers (diffusion-proof), sampling methods (not surface samples) and handling (store cool and dark, deliver to laboratory within 24 hours and rapid extraction at the lab) should be used.

For excavation of organic contamination to be delivered to soil treatment plants, the number of required samples primarily depends on the need to sort the soil into different categories with a view to achieving financial advantages. Therefore, the number depends on the specific case (amount, homogeneity, type of contamination, price differences between categories). Different clients, environmental authorities, and soil treatment plants can have their own requirements for the number of analyses. The final environmental documentation should be carried out when the soil is deposited after treatment.

Assigning soil directly to a landfill requires greater certainty and therefore a larger number of samples than soil assigned to a treatment plant. The number of samples and methods of analysis depend on the specific case (amount, homogeneity, type of contamination, final landfill), and the environmental authorities' requirements for documentation. The number of samples will typically be 1 sample per 30 tonnes when the soil is excavated due to contamination.

Requirements for the selection of analysis parameters and methods are the same as in the investigation phase and depend on the type of contamination. These are described in Appendix 4.9. Analyses must be carried out at an accredited laboratory. Detection limits must, as a rule, be 1/10 of the quality criteria.

For organic contamination, it should be noted that the content of heavy metals in the excavated soil (particularly lead) should also be known, since a high content of metals may be crucial for the treatment and the price.

At the receiving treatment facility, samples are often routinely analysed as part of the receiving and registration procedure. There may be variances in analysis results compared with those carried out by the supervision authorities. Therefore, it is important to note where samples have been collected.

Contamination investigations endeavour to determine the extent of the contamination so that the amount of soil to be excavated can be ascertained. The detailed project almost always provides an estimate of the amount of contaminated soil. There are often inconsistencies between the estimated amounts and the actual amounts because investigations are carried out based on spot checks. The actual amount excavated is obtained from weight notes completed when the soil is transported. In order to avoid misunderstandings, the supervisory inspection prepares forms that specify the load. The form should be signed by the driver, the supervision personnel, and the receiving facility, and then returned to the supervision personnel.

10.2.2 Documentation of residual contamination

Excavation is stopped temporarily when it is estimated by an inspection that adequate contamination has been excavated to have reached compliance with the previously set stop criteria. This compliance with the stop criteria should be documented by collecting an appropriate number of samples for chemical analysis from the sides and bottom of the excavation. These samples may be supplemented by field measurements. The number of samples should be agreed upon in advance with the supervision personnel. The focus should be on the most critical areas. If remediation is carried out due to risks involving outdoor areas and indoor air, most of the samples should be taken from the uppermost metres, while for risks involving the groundwater, there should be more documentation from the bottom of the excavation. The density of samples also depends on the nature of the contamination (is it visible, is it detectable using field methods, is the soil homogenous, etc.?).

Under the description of acceptance criteria in the detailed project, a minimum number of analyses must be stated, as well as the number of analyses per area unit. As a rule, samples should always be collected from all sides and the bottom of the excavation. Normally, as a minimum, sampling should correspond to level 3 as described in the guidelines on sampling and analysis of soil /3/. In cases where there are visual indications in the open excavation that the contamination distribution is inhomogeneous, for example through the distribution of geological layers and/or colour, more samples should be taken for residual contamination. The samples should be handled as described in the section above.

If contaminated soil remains in areas not accessible by the excavator, for example under buildings, a risk assessment should determine which precautions should be taken. For further details, refer to Chapter 5.

To document completion of the project, a report is prepared which documents that agreements have been complied with, including excavation, handling, and analysis procedures, and concentration levels in both excavated soil and remaining soil. A risk assessment should be carried out to determine the consequences of allowing residual contamination to remain.

10.3 Evaluation of in-situ remediation of soil contamination

Running-in and operation of remediation systems are described in Chapter 9. The following is a description of the concepts of evaluating the operation and final compliance. Evaluation of operation takes place at regular intervals during operation with a view to ascertaining the progress of remediation and whether the technical equipment is working optimally. The final compliance takes place when an evaluation of operation shows that it is likely that the acceptance criteria have been reached.

10.3.1 Evaluation of the operating period

The following describes possible evaluations for in-situ remediation methods used for soil contamination, including active methods such as soil vapour extraction and bioventilation, and passive methods such as immobilisation methods.

During the operating period, the contamination should be monitored so that changes in the contamination can be documented. In soil vapour extraction, operation should initially be followed closely (samples should be collected at least one week after the start), and subsequently with increasing time intervals, for example after 1, 3, 6, 9, and 12 months. Further evaluation of operation after this period can be set according to the results of the first year's operation /32/. This will typically be about 2-4 times each year.

During the operating period, measurements will primarily be of the contaminants in the discharged air. Furthermore, the air flow and the air pressure should be monitored. As completion of the remediation approaches, monitor wells for soil gas/groundwater may be included in the evaluation. In addition to measuring contaminants, it is possible to measure for oxygen, carbon dioxide, and temperature in the discharged air. The final evaluation of the success of the remediation is described in Section 10.3.2.

In bioventilation, evaluation of the operating period is best carried out by measuring oxygen and carbon dioxide consumption using bio-activity tests. By comparing with earlier measurements, an indication of changes in activity is obtained. In bio-activity tests, a specific quantity of oxygen is injected into the contaminated layer. Changes in the oxygen and carbon dioxide content are subsequently measured in the nearby monitoring wells. As a minimum, this should be carried out twice a year.

In addition, contaminants in the soil gas should be analysed for in existing monitor wells at the end of the operating period. Monitoring groundwater/soil can also be advantageous with regard to checking whether the water/soil is cleaner. Air measurements will typically be performed if the stop criteria consists of air concentrations. The same applies to water/soil. In monitoring groundwater, the redox conditions should be checked in order to determine the degradation potential.

In forced leaching, evaluation of the operating period is best performed by analysing water samples from the inlet and outlet of the water-treatment device. Analyses should be carried out an appropriate number of times at increasing time intervals. This should be done about once a week in the first month and subsequently about once a month, extending to a minimum of once every six months.

In addition, samples from monitor wells should be analysed at appropriate intervals during the operating period for content of contaminants in the groundwater aquifer.

In immobilisation methods (fixing/capping methods) the clean side of the cut-off system should initially be monitored twice a year, falling to once a year. For volatile contamination, it is normal to carry out soil gas measurements, while for water soluble substances, the groundwater aquifer should be monitored downgradient of the cut-off system. For extra security and to achieve optimal monitoring, it may be relevant to construct double walls, with monitor wells placed between the walls.

10.3.2 Stop criteria and the final compliance

Stop criteria should be set before commencement of remedial measures. The following parameters should be included in the decision process:

- Sample medium for final compliance (air/soil/water, possible combination)
- Procedures for evaluating final compliance
- Strategy of sampling to determine the lasting effect of remediation
- Measurement and analysis parameters
- Measurement and analysis procedures
- Permissible variation in results

Contamination may be in the water, soil, and/or air phase. Therefore, it is possible to determine changes in the contamination in a single medium, or in a combination of several media. For example, if the indoor air is threatened, a soil gas criterion could be set. If remediation takes place due to risks for outdoor areas, a soil criterion may be appropriate, and a groundwater criterion could be set if drinking water is threatened. It is often necessary to establish new wells/boreholes between existing wells/boreholes.

The normal procedure for evaluating final compliance is to obtain a concentration in the media (soil, water, air) which relates to the limit values for individual substances. In a few cases, the obtained ratios between individual contaminants have been used. In these methods, substances which are quickly removed are compared with substances which are difficult to degrade /46/. This method can only be used in remediation where the substances which degrade rapidly are the most critical, usually for indoor air problems.

Finally, stop criteria may be interpreted pragmatically in relation to the rate of remediation. When the remediation process is sufficiently slow, remedial measures can be stopped temporarily. a subsequent risk assessment forms the basis for deciding whether remedial measures should be stopped permanently, or whether it is necessary to continue using another technique. Several remediations have followed this process in practice.

There is a difference between the effect during operation and the lasting effect for many in-situ remedial methods. In some cases, contaminants will reappear/flow back after equipment is shut off, causing the remediation to fail to comply with original stop criteria (rebound effect). The final compliance evaluation should therefore establish a sampling strategy which will document the lasting effect (that the remediation is satisfactory). Thus, before commencing remedial measures, decisions should be made on which sample medium will provide the best evidence of the lasting effect, the number of samples required for sound decision-making (when it absolutely certain that the remediation effect is lasting), the length of time between sampling, and whether individual samples or mixed samples (soil) should be taken. Typically, there should be assessments of how long it will take before contamination is transported into the water phase and further into compliance wells.

When the evaluations during the operating period reveal adequately low concentrations in air discharge (where this is the only requirement for operation), samples must be collected from the sample medium selected for the final compliance. Samples must be collected from places other than the air discharge for ventilation methods. As a minimum, two consecutive analyses of the air discharge should show no measurable contamination. The samples should be collected with about a two month interval, where the pump has been stopped for a period. Following this, samples of soil/water/air can be collected from new wells in order to check for compliance with stop criteria.

In forced leaching, the quality criteria for compliance are fixed concentrations either in soil or in groundwater. However, in practice, these pre-set criteria have not been used. For in-situ remediation completed up to now, risk assessment of residual contamination has provided the basis for stopping remediation.

For most passive in-situ methods, e.g. immobilisation, there is no distinction between final compliance and evaluating the operating period. Monitoring corresponding to evaluating the operating period is continued (although as time goes by, longer intervals between monitoring rounds are used).

Decisions must be made on which contaminants are to be quantified through analyses. These may be individual substances or mixtures of substances. It is important to define the evaluation procedures to be used, including methods of analysis. Methods of analysis are described in the Guidelines on sampling and analysis of soil /3/. If a specific correlation is to be demonstrated, it may also be relevant to use field measurements as part of the stop criteria.

Before starting remedial measures, rules for interpreting variations in analysis results must be established. For example, there may be requirements that a specific percentage of the results must comply with criteria, while at the same time establishing a maximum concentration which no single analysis may exceed.

In in-situ remediation in Denmark, both air and water have been used as media for stop criteria. For soil vapour extraction, a number of cases have been completed where the stop criteria have been fulfilled. In forced leaching, there are examples of full remediation of both soil and water. A few remediations have been completed because the stop criteria have been achieved, but several remediations have been stopped due to operational problems.

10.4 Evaluating groundwater remediation

There are various methods/principles for remediation of groundwater, cf. descriptions in Chapter 9. The methods may be roughly divided into pump-and-treat and other in-situ techniques.

10.4.1 Evaluating pump-and-treat

When pumping is started, it is necessary to check whether the contamination is under hydraulic control. A monitoring programme is usually established when the plant is constructed, and this is revised once the installation has been run in. The monitoring programme lays down where the potentiometric surface should be measured and where measurements of pump yield are to be made, as well as how often the yield is to be measured.

Wells for monitoring hydraulic control are usually located within and near the borders of the capture zone of the well. Therefore, observations on both sides of the groundwater divide should document that the contamination plume is on the right side of the divide.

Evaluation of whether contamination has been remediated as planned includes sampling and analyses of contaminants in the pumping well and monitor wells. Wells for monitoring remediation should be located within the contamination plume, at the source of contamination, and possibly in minor upper aquifers above the contamination plume, downgradient of the source.

The monitoring programme establishes where samples are to be collected, and how often water samples are to be collected, as well as which analyses are to be conducted.

In pump-and-treat, there is often a difference between the remediation effects during operation and the lasting effects. When the pump is turned off, contaminants will often be released/flow back, so that remediation does not comply with the stop criteria as expected (rebound effect). For example, contaminating substances can be released into groundwater when the groundwater table rises after pumping is slowed down. It is therefore very important that the lasting effect is measured after pumping has been stopped. There should be estimates of when it will be possible to ascertain any rebound effects in wells. If the lasting effect is deemed to have been met, but control measurements after a period of 3 months or so indicate that stop criteria have been exceeded, pumping must be recommenced. This procedure should be repeated until the stop criteria have been achieved.

The desired remediation level is established on the basis of the risk assessment conducted in the investigation phase. The stop criteria should contain requirements that values below the remediation level are achieved for several consecutive monitoring rounds. In addition, samples should be analysed from several monitor wells as well as from the pump well. Stop criteria may be varied according to the location of the well from which the sample was collected.

Experience with the pump-and-treat method is great. Many projects have been completed. However, many projects have demonstrated difficulties in achieving the stop criteria. On the other hand, using hydraulic control, it is possible to prevent groundwater contamination from spreading towards water abstraction wells etc., and at the same time remove some of the contamination.

There are examples where successful remediation in relation to land use has occurred in aquifers near the surface. On the other hand, there are only a few examples of completed pump-and-treat projects in aquifers with high yields where the goal is to remediate to drinking-water standards.

Requirements for discharge are laid down in a discharge permit by supervision authorities for treatment plants, surface water recipients etc. In addition to requirements for parameters to be analysed, analysis procedures to be used, compliance concentrations, and permitted discharge amounts, a discharge permit also contains requirements on frequency of sampling and analyses. Therefore, in accordance with the required frequency of analysis, water samples must be collected prior to discharge into the sewer mains in order to check for compliance.

When discharging contaminated groundwater, evaluations should be made to ensure that the treatment processes are running satisfactorily. For example, in filter technology, all filters must be regularly backwashed, cleaned, or replaced. The effect of the filter is reduced over time as the filter material slowly loses its ability to ad/absorb components, or it becomes clogged. Therefore, a certain amount of monitoring and evaluation of operation must be expected in connection with backwashing, cleaning, and replacement of filters.

The extent of monitoring water treatment is very method specific, and should therefore be described in the monitoring programme. For example, separators must be emptied at appropriate intervals.

For water treatment with activated carbon, the system usually comprises two filters in series. The treatment effect of the system is best measured between the filters so that the filters can regularly be replaced one at a time, and contaminants never break through the final filter.

10.4.2 Evaluating in-situ remedial methods

Evaluation of the remediation effect of air sparging, as with pump-and-treat, is primarily done through monitoring the groundwater. This implies analysis of groundwater samples from monitor wells located centrally and on the periphery of the contaminated area. The frequency of sampling can for example be after 1, 2, 3, 6, 12, 18, and 24 months /32/. It should be noted that air sparging can cause significant spreading of contamination by transport in the sparged air in the saturated zone, probably as a result of low-permeability horizontal zones. The existence of such low-permeable zones should be examined in the design phase. If these zones are present, monitoring should also be carried out further away from the sparge area, in regard to possible indoor air problems as well as to groundwater. At the same time, it is also important to regularly measure the potentiometric surface of the groundwater in order to monitor mounding of the water table.

Concurrently with air sparging, it is normal to remove stripped contaminants from the unsaturated zone using soil vapour extraction. Measurements of these air emissions should be included in the monitoring programme.

For air sparging, the pragmatic view will generally apply with regard to stop criteria. Remedial measures can be stopped when concentrations of contaminants are low and there are no notable changes in the contamination pattern (even after taking possible rebound effects into account). In addition, a specific risk analysis should conclude that remedial measures can be stopped. Sparging may be replaced by monitoring or pump-and-treat if the risk analysis deems this necessary.

Reactive permeable barriers allow the passage of groundwater, while degrading or removing contamination from the groundwater. Compliance is evaluated in the groundwater zone and should include samples collected before inflow, in the barrier itself, and after the barrier. Furthermore, contamination should be monitored upgradient and downgradient, as well as before and after any cut-off walls in order to examine effectiveness. To ensure the required flow direction, and in order to enable commencement of possible measures against mounding problems, the groundwater table should also be monitored.

Impermeable barriers should primarily be monitored downgradient of the contamination. As an extra measure to achieve optimal evaluation, it may be relevant to establish double barriers with monitoring between the barriers.

With methods where oxidising agents are added to the groundwater zone, effectiveness should be evaluated downgradient and in pump wells where the purpose is to produce an ‘oxygen barrier’. In addition to contaminants, monitoring should identify when the oxidising agent should be replaced or recharged.

10.5 Evaluating soil gas remediation

In buildings where there is a risk of indoor air problems, and where remedial measures have been started as a result of this risk, the effect of these remedial measures should be evaluated at regular intervals. This should usually be carried out by taking measurements under the floor, and not in the building itself, so that false sources can be ignored. If measurements are conducted of the indoor air, and not of the soil gas, reference samples should always be taken in rooms which are not affected by the contamination. Background measurements should also be taken outside.

In cases where passive or active ventilation has been established, for example ventilated drains in the capillary-breaking drainage layer, evaluation can be carried out by measuring emissions to the air or by taking measurements in ventilation pipes under the floor. If contamination cannot be detected in the air discharged, the equipment can be stopped and final evaluation can subsequently be carried out by measurements in the soil gas (possibly in drainage pipes). Active ventilation can be stopped when contents less than the stop criteria have been recorded at least twice at intervals of two months. It should be noted that after stopping pumps, rebound effects may occur in drainage pipes, and therefore it is never sufficient to conduct a single round of measurements.

10.6 Controlling measures for landfill gas

A monitoring programme should be established at landfills where gas generation from rubbish presents a risk to people or the environment. A suggested monitoring programme for landfill gas is described below. For further information, refer to a report from the Danish Environmental Protection Agency on landfill gas /17/.

The objective of monitoring is to check changes in the amount of gas in order to determine whether remedial measures should be started or changed, or whether remedial measures in progress continue to be satisfactory.

Landfill gas can be monitored at several locations, including:

- On the surface
- Underground with soil lances
- In monitoring screens installed in excavated ditches
- In monitor wells
- In existing wells (leachate collection wells, etc.)
- In buildings equipped with alarm systems

The most common method is measurement in monitor wells or soil lances. Location and intervals of lances or wells are determined from surveys of the risks of gas generation and gas flow. Location depends on the methane content, the amount of gas, gas penetration through the rubbish, the extent of the landfill, the surrounding geological strata, and the distance to buildings, pipe systems, and sewers. Finally, the location depends on the design of the remedial installation.

There should always be measuring points outside the extent of the landfill, particularly between the landfill and any nearby buildings. A more detailed description of the design of wells and their location appears in a report from the Danish Environmental Protection Agency /17/. An example of the design of a stationary measuring point appears in Appendix 4.6.

Excavated ditches can be utilised at shallow landfills. Direct push or hammered lances are mostly used for point sources due to their limited range. Measurements at existing water supply wells and leachate monitoring wells can be used to supplement, but not replace specially constructed gas-measuring points.

Where remedial measures have been completed for landfill gas, measurements should if possible be made between the preventive installations and buildings using the above monitoring sites. This is possible if gas barriers, permeable ditches, ventilation drains, or wells are used as remedial measures between the source of gas and the building. If landfill gas is not recorded from repeated monitoring in wells, the frequency of monitoring can be reduced. In cases where gas penetration is cut-off using technical construction measures or by changing the pressure gradient in the building, monitoring should primarily be carried out under the floor or inside the building.

In buildings in high-risk areas, systems for measuring gas concentrations and gas alarms should be installed as an extra check to ensure the effectiveness of remedial measures. In cases of a methane content of more than 1 per cent by volume, defined as 20 per cent of the lower explosion limit of 5 per cent methane by volume, or in cases where there is a carbon dioxide content of more than 5 per cent by volume, the building should be evacuated /17/. There should always be an action plan for the safety of people in the building, and everyone should be acquainted with this plan.

Monitoring should continue until the risk to people and the environment has abated; that is, until gas concentrations are not explosive (methane) or toxic (carbon dioxide). Concentrations of methane must be less than 1 per cent by volume, and concentrations of carbon dioxide from decaying waste must be less than 1.5 per cent by volume, as measured inside the landfill over a longer period. For example, this may be a two-year period where measurements are taken under different weather conditions at least every six months (warm weather in summer and frozen soil in winter, as well as during periods with decreasing atmospheric pressures while below an absolute value of 1,000 millibars). Alternatively, an actual investigation of the waste could be performed and provide statistical certainty that the degradable waste has been degraded /17/.

Consideration should be given to the safety of people who are employed to establish evaluation and monitoring. Therefore instructions for procedures regarding safety at work should always be available.

11 References

/1/ Vejledning om kortlægning af jordforurening og kilder hertil. *(Guidelines on mapping soil contamination and sources*). Draft Guidelines, 1997, Danish Environmental Protection Agency.

/2/ Projektering. Course for Project Managers 2, 6 April 1997. Amternes Depotenhed.

/3/ Vejledning om prøvetagning og analyse af jord. (*Guidelines on sampling and soil analysis*) Draft Guidelines, 1997, Danish Environmental Protection Agency.

/4/ Vejledning om rådgivning af beboer i let forurenede områder (*Guidelines on informing residents living in slightly contaminated areas*) Draft Guidelines, 1997, Danish Environmental Protection Agency.

/5/ Kilder til industrikortlægning (*Sources of industry mapping*) Elucidation Report U6. Landfill Project 1989.

/6/ Statutory Order no.4 of 4 January 1980 on execution of groundwater wells. Ministry of the Environment.

/7/ Vejledning i måling af stoffer i indeluften fra forurening i jorden (*Guidelines for measurement of substances in indoor air from contamination in soil*) Building and Housing Agency 1994.

/8/ Vejledning i ingeniørgeologisk prøvebeskrivelse (*Guidelines on descriptions of geological samples*) Danish Geotechnic Association Bulletin 1 July 1988.

/9/ Toksikologiske kvalitetskriterier for jord og drikkevand (*Toxicological quality criteria for soil and drinking water*) Danish Environmental Protection Agency Report no. 12 1995.

/10/ Økotoksikologiske jordkvalitetskriterier. (*Ecotoxicological soil quality criteria*) Danish Environmental Protection Agency Report no. 13 1995.

/11/ B-værdier (*B values*). Orientation from the Danish EPA, Orientering no. 15, 1996.

/12/ Risikovurdering af forurenede grunde (*Risk assessment of contaminated sites*). Environmental Project no. 123. Danish EPA, 1990.

/13/ Kemiske stoffers opførsel i jord og grundvand (*Behaviour of chemical substances in soil and groundwater*). Project no. 20, Danish EPA, 1996.

/14/ Radon og nybyggeri (*Radon and new construction*). Building and Housing Agency, 2nd edition, 1993.

/15/ Baggrundsværdier for organiske forbindelser i indeluften (*Background values for organic compounds in indoor air*). Building and Housing Agency, 1994.

/16/ Undersøgelse af lufttæthed i bygningskonstruktioner (*Survey of air-tightness of building constructions*). Building and Housing Agency, 1995.

/17/ Losseplads gas (*Landfill gas*). Working Report from the Danish EPA no. 69, 1993.

/18/ Nielsen, G.D. et al, Flygtige organiske forbindelser i indeluft – Stoffer, koncentration og vurdering (*Volatile organic compounds in indoor air*). Dansk Kemi 4, 1997.

/19/ Begrænsning af luftforurening fra virksomheder (*Limiting air pollution from enterprises*) Guidelines from the Danish EPA no. 6, 1990.

/20/ Acceptkriterier for mikrobiologisk renset jord (*Acceptance criteria for microbiologically remediated soil*). Guidelines from the Danish EPA no. 8, 1992.

/21/ B-værdier (B values). Orientation from the Danish EPA no. 15, 1996.

/22/ General branchevejledning for forurenede grunde (*General sector guidelines for contaminated sites*) Guidelines from the Danish EPA no. 5, 1992.

/23/ Branchevejledning for forurenende træimprægneringsgrunde (*Sector guidelines for contaminated wood-treatment sites*) Guidelines from the Danish EPA no. 4, 1992.

/24/ Branchevejledning for forurenede garverigrunde (*Sector guidelines for contaminated tannery sites*). Guidelines from the Danish EPA no. 5, 1992.

/25/ Branchevejledning for forurenede tjære/asfaltgrunde (*Sector guidelines for contaminated tar/asphalt sites*). Guidelines from the Danish EPA no. 6, 1992.

/26/ Boringskontrol på vandværker (*Well control at water works*). Guidelines from the Danish EPA no. 2, 1997.

/27/ Projektstyringshåndbog for jord- og grundvandsforureninger (*Project management handbook for soil and groundwater contamination*). Soil and groundwater project, Danish EPA no. 5, 1995.

/28/ Act No. 216 of 8 June 1966 on tendering etc.

/29/ Statutory Order no. 1017 of 15 December 1993 on organisation of construction sites and similar workplaces. Ministry of Labour.

/30/ AB92. (*Normal conditions for work and supply at building and construction enterprises*) Ministry of Housing, 10 December 1992.

/31/ Barrierer mod udvikling og anvendelse af nye afværgeteknologier (*Barriers to developing and using new remedial technology*). Soil and groundwater project, Danish EPA no. 4, 1995.

/32/ Erfaringer med in-situ afværgeforanstaltninger (*Experience with in-situ remedial measures*). Soil and groundwater project, Danish EPA no. 7, 1995.

/33/ DS 415. *Contamination Norms.* Danish Society of Engineers, 1984.

/34/ Forurenet og renset jord på Sjælland og Lolland Falster. Vejledning i håndtering og bortskaffelse (*Contaminated and treated soil on Zealand and Lolland Falster*). Local Authorities of Frederiksberg and Copenhagen, Counties of Frederiksborg, Roskilde, Storstrøm and West Zealand.

/35/ Beckett, G.D, Huntley, D.. Characterisation of Flow Parameters Controlling Soil Vapour Extraction. Groundwater, March-April 1994.

/36/ Hinchee, R.E., Ong. S.K., Miller, R. N., Downey, D. C., Frandt, R. Test Plan and Technical Protocol for a Field Treatability Test for Bioventing. US Air Force Center for Environmental Excellence, Brooks Air Force Base, Texas, USA.

/37/ Bioremediation of Contaminated Soil. Working Report Danish EPA no. 4, 1995.

/38/ Forureningsbekæmpelse med mikroorganismer (*Combating contamination with microorganisms*). Theme Report from the DMU, 1996.

/39/ Electrokinetic Remediation of Heavy Metal Polluted Soil. Working Report from the Danish EPA no. 67, 1994.

/40/ Design, indkøring og drift af afværgepumpning (*Design, running in and operation of remedial pumping*). Soil and groundwater project, Danish EPA no. 1, 1995.

/41/ Udnyttelse og rensning af forurenet grundvand (*Use and treatment of polluted groundwater*) Soil and groundwater project, Danish EPA no. 2, 1995.

/42/ DS 432. Norms for drainage installations. 2nd edition, 1994

/43/ Air sparging from horizontal soil borings. Working Report from the Danish EPA no. 9, 1997.

/44/ Gillham, R.W. In Situ Reactive Barriers for Plume Control. Diagnosis and Remediation of DNAPL Sites. Waterloo. DNAPL Course. ATV, Vingsted, 6-8 May 1996.

/45/ Restforureninger under huse. (*Residual contamination under houses*) Soil and groundwater project, Danish EPA no. 22, 1996.

/46/ L. Ramsay. Stopkriterier for in-situ oprensninger (Stop criteria for in-situ remediation). Vand og jord 1995.

/47/ Statutory Order no. 921 of 8 October 1996 on quality requirements for aquatic areas and requirements for emissions of certain hazardous substances to waterways, lakes, or the sea.

/48/ Statutory Order no. 310 of 25 April 1994 on waste water permits etc.

/49/ Statutory Order no. 75 of 30 January 1992 on limit values for emissions of certain hazardous substances to waterways, lakes and the sea (List 1 substances).

/50/ Statutory Order no. 181 of 25 March 1986 on limit values for cadmium with waste water from certain industrial plants.

/51/ Statutory Order no. 520 of 8 August 1986 on limit values for mercury in emissions from certain industrial plants.

/52/ Statutory Order no. 6 of 1974 from the Danish EPA. Guidelines for provisions on emissions of waste water.

/53/ Økotoksikologiske jordkvalitetskriterier (*Ecotoxicological soil-quality criteria*). Working Report from the Danish EPA, no. 83, 1997.

/54/ Diffusionsforsøg, betongulv (Diffusion tests, concrete floors). Danish Housing and Building Agency, 1992.

/55/ Modværdien i danske kroner af EU udbudsdirektivernes tærskelværdier for perioden januar 1996 til 31. December 1997. (*Equivalent values in DKK of threshold values in EU tender directives from January 1996 to December 1997*).

/56/ Kjærgaard, M., Ringsted, J.P., Albrechtsen, H.J. and Bjerg, P.L. 1998. Naturlig nedbrydning af miljøfremmede stoffer i jord og grundvand (*Natural degradation of xenobiotic substances in soil and Groundwater*). Report by the Geo-technical Institute and DTU for the Danish EPA.

/57/ Wiedemeier, T.H. et al. 1995. Technical Protocol for Implementing Intrinsic Remediation with Long-Term Monitoring for Natural Attenuation of Fuel Contamination Dissolved in Groundwater. Air Force Center for Environmental Excellence, Technology Transfer Division, Brooks Air Force Base, San Antonio, Texas.

/58/ Wiedemeier, T.H. et al. 1995. Technical Protocol for Evaluating Natural Attenuation of Chlorinated Solvents in Groundwater. Draft – revision 1, Air Force Center for Environmental Excellence, Technology Transfer Division, Brooks Air Force Base, San Antonio, Texas.

Appendices - Content

Production conditions for sectors of industry

References to background material

Type of enterprise	Reference
Tarworks and Asphalt Plants	Sector guidelines for contaminated tar/asphalt sites (1992) /1/ Erfaringsopsamling på amternes registreringsundersøgelser ('Findings from county investigations') (1988) /2/ Appendix 3.5
Vehicle repair shops	Erfaringsopsamling på amternes registreringsundersøgelser ('Findings from county investigations') (1995) /3/ Sector description for vehicle repair shops (1997) /16/
Dye works	Erfaringsopsamling på amternes registreringsundersøgelser ('Findings from county investigations') (1995) /3/
Galvanisation companies	Erfaringsopsamling på amternes registreringsundersøgelser ('Findings from county investigations') (1995) /3/ Contaminated industrial sites (1988) /2/
Tanneries	Sector guidelines for contaminated tannery sites (1992) /4/ Contaminated industrial sites (1988) /2/ Sector description for tanneries (1997) /17/ Appendix 3.4
Gas works	Contaminated gas-works sites /15/ Contaminated gas-works sites (1989) /5/ Contaminated industrial sites (1988) /2/
Industrial paint shops	Erfaringsopsamling på amternes registreringsundersøgelser ('Findings from county investigations') (1995) /3/
Iron and metal foundries	Historical description of iron and metal foundries' possible environmental impact (1992) /6/ Contaminated industrial sites (1988) /2/ Sector description for iron and metal foundries (1997) /18/
Grain and feed companies	Historical description of grain and feed companies' possible environmental impact (1992) /7/ Sector description for grain and feed enterprises (1997) /19/
Varnish and Paint plants	Contaminated industrial sites (1988) /2/

Type of enterprise	Reference
Machinery works	Erfaringsopsamling på amternes registreringsundersøgelser (‘Findings from county investigations’) (1995) /3/ Historical description of the machinery sector’s possible environmental impact (1992) /8/ Sector description for metal processing plants (1997) /20/
Scrapyards, car breakers, etc.	Sector description for scrapyards, car breakers and iron and metal recovery enterprises (1997) /21/
Plastic plants	Historical description of the plastic sector’s possible environmental impact (1992) /9/
Dry cleaners	Erfaringsopsamling på amternes registreringsundersøgelser (‘Findings from county investigations’) (1995) /3/ Historical description of the dry cleaning sector’s possible environmental impact (1992) /10/
Service stations	Appendix 3.6
Roofing-felt companies	Sector guidelines for contaminated tar/asphalt sites (1992) /1/ Contaminated industrial sites (1988) /2/ Appendix 3.5
Printing companies	Historical description of the printing sector’s possible environmental impact (1992) /12/
Wood and furniture companies	Erfaringsopsamling på amternes registreringsundersøgelser (‘Findings from county investigations’) (1995) /13/ Historical description of the wood and furniture sector’s possible environmental impact (1992) /12/
Wood preservation companies	Sector guidelines for contaminated wood preservation sites (1992) /13/ Contaminated industrial sites (1988) /2/ Sector description for wood preservation companies (1997) /22/ Appendix 3.3
Heating plants	Sector description for heating plants (1997) /23/
Vulcanising plants	Historical description of vulcanising plants’ possible environmental impact (1992) /14/

References

/1/ Sector guidelines for contaminated tar/asphalt sites
Guidelines from the Environmental Protection Agency No. 6, 1992

/2/ Contaminated industrial sites
Report U2
The Landfill Project, 1988.

/3/ *Erfaringsopsamling på amternes registreringsundersøgelser* (‘Findings from county investigations’)
Project on soil and groundwater from the Environmental Protection Agency No. 9, 1995.

/4/ Sector guidelines for contaminated tannery sites
Guidelines from the Environmental Protection Agency No. 5, 1992.

/5/ Contaminated gas-works sites
Report U4
The Landfill Project, 1989.

/6/ Historical description of Iron and Metal Foundries’ possible environmental impact – with particular reference to soil and groundwater contamination . Vestsjællands Amtskommune.
Teknisk Forvaltning. 1992.

/7/ Historical description of Grain and Feed companies’ possible environmental impact – with particular reference to soil and groundwater contamination . Vestsjællands Amtskommune.
Teknisk Forvaltning. 1992.

/8/ Historical description of the Machinery sector’s possible environmental impact – with particular reference to soil and groundwater contamination . Vestsjællands Amtskommune.
Teknisk Forvaltning. 1992.

/9/ Historical description of the Plastic sector’s possible environmental impact – with particular reference to soil and groundwater contamination . Vestsjællands Amtskommune.
Teknisk Forvaltning. 1992.

/10/ Historical description of the Dry Cleaning sector’s possible environmental impact – with particular reference to soil and groundwater contamination . Vestsjællands Amtskommune.
Teknisk Forvaltning. 1992.

/11/ Historical description of the Printing sector’s possible environmental impact – with particular reference to soil and groundwater contamination . Vestsjællands Amtskommune.
Teknisk Forvaltning. 1992.

/12/ Historical description of the Wood and Furniture sector’s possible environmental impact – with particular reference to soil and groundwater contamination . Vestsjællands Amtskommune.
Teknisk Forvaltning. 1992.

/13/ Sector guidelines for contaminated wood preservation sites.
Guidelines from the Environmental Protection Agency No. 4, 1992.

/14/ Historical description of Vulcanisation plants’ possible environmental impact – with particular reference to soil and groundwater contamination . Vestsjællands Amtskommune.
Teknisk Forvaltning. 1992.

/15/ Contaminated gas work sites. Meeting, 9 October 1996. The committee on groundwater contamination.

/16/ Sector description for vehicle repair plants.
Amternes Depotenhed. Teknik & Administration, No. 4, 1997.

/17/ Sector description for tanneries. Amternes Depotenhed. Teknik & Administration, No. 5, 1997.
/18/ Sector description for iron and metal foundries. Amternes Depotenhed. Teknik & Administration, No. 6, 1997.
/19/ Sector description for grain and feed companies. Amternes Depotenhed. Teknik & Administration, No. 7, 1997.
/20/ Sector description for metal processing plants. Amternes Depotenhed. Teknik & Administration, No. 8, 1997.
/21/ Sector description for Scrapyards, car breakers, and metal recovery companies. Amternes Depotenhed. Teknik & Administration, No. 9, 1997.
/22/ Sector description for wood preservation companies. Amternes Depotenhed. Teknik & Administration, No. 10, 1997.
/23/ Sector description for heating plants. Amternes Depotenhed. Teknik & Administration, No. 11, 1997.

Check list for site visit

Use a large scale site plan (e.g. 1:200)
Remember to take photographs of the site.

Installation/activity	Observations
Site plan	Are scale and compass arrow accurate?
Present land use Previous land use	Buildings, activities, washing facilities, storage, etc. Later additional buildings (if applicable)
Land use of adjacent sites	Buildings, activities, storage, etc.
Ventilation nozzles/flanges	Number, location
Tanks/covers	Location Tank No. Datum level of tank bottom Volume/diameter Spillage at filler connection Product Product amount in reservoir
Oil/petrol separators	Condition/location
Sewerage and drain	Location, signs of contamination
Pumping islands/other installations	Location, signs of contamination Look for concrete foundations or other types of paving (at individual stand)
Other signs of contamination	Discoloration Odours Damage to vegetation
Buildings	Construction/Condition Basement Foundation Washing hall Greasing pit etc.
Paving	Types Signs of breakage/new asphalt Concrete underneath asphalt Is breakage necessary by field work?
Access	Shed roofs and sheds Gates and fences Storage Trees, shrubbery, etc. Overhead lines
Topography	Estimated flow direction of aquifers near ground level Alterations to the terrain
Nearby recipients	Oil film
Other information	Interviews with site owners etc.

Instructions for carrying out borings and wells.

This Appendix describes how to carry out borings and wells conducted in connection with contamination investigations. In addition to drilling, screen installation and grouting the well casing is also covered. The following points are addressed:

- Drilling
- Maintaining records
- Screen installation
- Grouting well casing
- Measuring-in and levelling

Soil sampling is covered in Appendix 4.2, water sampling in Appendix 4.3. It has been the intention to develop the following as a set of minimum requirements that should be met when drilling. As a consequence, this Appendix can be included as the basic requirements in connection with submission of tenders for drilling and well construction. Additional requirements must be formulated in relation to the purpose and nature of individual investigations.

Statutory Order No. 4 (4 January 1980) on the execution of wells for groundwater /3/ specifies present regulations for drilling, abandoning, construction, and notification, etc. According to the Statutory Order on design of wells for groundwater, GEUS must be notified within three months of drilling of any wells serving this purpose.

The designation ’localisation borings’ is used to denote soil borings conducted in order to identify and describe contamination of the upper soil strata and groundwater aquifers near ground level. Such borings are often used during the preliminary stages of the investigation process.

In connection with very shallow soil borings (1-2 m) , hand drilling is possible. For this purpose a post-hole auger or a spade drill can be used (Dutch drilling equipment) /1/. Otherwise, shallow exploratory soil borings are usually carried out using a solid-stem auger or a bucket auger as drilling tool. If geological conditions permit, the borings can be executed without the use of casing, since this method can provide adequate level-specific soil samples in connection with shallow borings (to a depth of approximately 3 m and above groundwater level).

In principle, the dimension of the borehole is discretionary. The choice of dimensions must be consistent with any desired screen size, including the gravel pack. Normally, a screen with a diameter of 63 mm (50 mm) is inserted in 6" soil borings, whereas only a 25 mm piezometer can be placed in

4" soil borings. If screens are installed in an actual groundwater aquifer, the well casing should always be grouted with bentonite, typically at the level of any low-permeable strata. Figure 1 features an example of a screen installation.

Shallow localisation soil borings can be filled back with the excavated material if the soil borings do not exceed a depth of 3-4 m, or if they do not cut through more than one groundwater aquifer.

In contrast to the shallow localisation soil borings, casing should always be used during the drilling of investigation borings. This is due to the fact that these borings are deeper, and that satisfactory sampling is usually the objective, either of soil or water or both. The use of casing will prevent cross contamination between the various strata. Also, the use of casing is necessary when drilling below the groundwater level

Normally, augering is used. However, with non-cohesive soil (sand/gravel) below groundwater level, the cable tool method will often be used /1/. When using a cable tool, water is usually added. It is necessary to be aware of this in connection with subsequent water-sampling, where removal of stagnant water is particularly important to facilitate representative sampling.

Other drilling techniques may be appropriate to the extent that no soil samples are required or that an impaired quality and/or quantity of geological information is acceptable. Such techniques might for include e.g. driven well or drilling with a hollow-stem auger. Deep investigation borings should not be located in the most contaminated areas (hot spots) of a contaminated site /2/.

One of the primary targets of investigation soil borings is to ensure the water sampling over a long period, at specified depths, through a screen. As a consequence, the dimensions of soil borings must be selected according to the desired screen installation. Screen dimensions are selected according to the purpose of the soil boring. Typical associated dimensions and maximum pumping yields are presented in Table 1.

Table 1
List of recommended screen dimensions

Purpose	Minimum screen diameter/ mm
Field measurements or sampling with suction pump etc.	25 (21.5 mm inside)
Sampling with smallest submersible pump (max yield approx. 2 m^3/h)	63 (52 mm inside)
Pumping with smaller submersible pumps (max yield approx. 15 m^3/h)	110 (99.4 mm inside)
Pumping with larger submersible pumps (max yield approx. 15-40 m^3/h)	160 (149 mm inside)
Pumping with larger submersible pump (max yield approx. 40-80 m^3/h)	225 (203 mm inside)
Pumping with largest submersible pump (max yield approx. 40-80 m^3/h)	315 (285 mm inside)

The connection between the above maximum yield values and the associated minimum dimensions must be seen as guidelines. The above values are determined using the dimensions for commonly used submersible pumps (such as GRUNDFOS type). Attention must be drawn to the fact that pump types exist with different dimensions, and to the fact that the water head influences pump yield.

Table 2 presents guidelines for the connection between borehole dimensions and screen dimensions. The recommended screen dimensions should not be exceeded, as this will result in and insufficient gravel pack and grouting.

The borings must be protected at ground level by a concrete protective casing with a cap, or by installing dry wells. Wells to deeper groundwater aquifers must be fitted with lockable seal caps or covers.

Table 2
List of recommended screen dimensions

Borehole diameter	Screen diameter/ mm
4" = 100 mm	25, (50)
6" = 150 mm	63, (90)
8" = 200 mm	110, (125)
10" = 250 mm	160, (140)
12" = 300 mm	200, (160, 225)
16" = 400 mm	250, (315)

The designation ’wells’ denotes borings whose construction correspond to water-supply wells. The primary purpose of these borings is to pump up groundwater from a groundwater aquifer, whereas soil samples from these borings are only used for geological assessment and are analysed only if contamination is registered during boring. Borings of this kind are primarily used for monitoring groundwater, and sometimes for remedial pumping. With the exception of remedial borings, wells should not be located in the most contaminated areas (hot spots) of a contaminated site /2/.

When carrying out control wells outside the contaminated site, there is basically freedom of choice as regards drilling method. This means that rotary drilling or air drilling, e.g. Odex drilling might be considered. These methods can be advantageous in connection with solid rock such as carbonate deposits. /1/. When carrying out rotary drilling, where drilling fluid is added to the borehole, the chemical content of the fluid should be tested. It is also particularly important to carefully remove the drilling fluid by developing the well before water sampling takes place. Rotation drilling should only be utilised outside the contaminated site.

The size of the borehole is chosen in accordance with the intended screen dimension. With these types of borings, a relatively large screen dimension will be selected, such as 160 mm, 200 mm, or 250 mm. See Table 2.

The borings must have a concrete protective casing as the minimum protection. It may prove necessary to establish an actual dry well or, as an alternative, an insulated covering at ground level. Wells reaching lower groundwater aquifers must be reported to GEUS. Table 3 has an overview over recommended drilling methods.

Table 3
List of recommended drilling methods

	Localisation soil borings	Investigation borings	Wells
Augering	x	x	x
Cable tool		x	x
Electrical log soil boring	x	x	
Hollow-stem auger	x	x	
Driven soil borings	x		
Air drilling			x
Rotation drilling			x

During drilling, a record should be maintained stating:

- Site, soil boring no., and date
- Drilling method
- Preliminary soil type assessment/classification
- Recorded signs of contamination (discoloration, odour)
- Stratum boundaries
- Sampling depth
- Drilling depth
- Screen installation
- Grouting
- Water level observations

The screen installation should be described in a scale drawing. Water level observations made during soil boring should be indicated in relation to a permanent measuring point that can be found again later.

Normally, threaded PEH or PVC tubes are used as observation and screen pipes. The thread facilitates oil-free pipe connection.

Screen installation is carried out by means of slotted tubes packed with gravel. During screen installation, it can be expedient to place a length of casing to collect fine-grained material – a silt box.

The screen arrangement must be closed at the bottom using a PVC or wooden plug (not pressure-impregnated wood).

To account for possible blocking around the casing, soil borings that penetrate limestone rock or other hard rock must also have screens installed. The screen must be established near the aquifer layer. The screen installation should represent only one groundwater aquifer.

The screen can comprise the entire groundwater aquifer or parts thereof. By using short screens, low dilution and more depth-specific samples are obtained. However, this method also means that screens will normally be required at more levels, hence more samples for analysis will be necessary. In localisation borings where no real aquifer layer has been found, screens of 1-2 m are installed from the bottom of the boring.

By screens in connection with unconfined groundwater tables, the upper side of the screen should be above the water table to facilitate registration of any oil or similar substances floating on the water table as well as taking account of fluctuations. The screen must be packed with cleansed quartz silica sand. The gravel packing must be carefully monitored to ensure that all cavities between screen and wall of soil boring are filled in. This monitoring must ensure that the upper side of the gravel packing constitutes a solid foundation for the necessary grouting in the annular space between well casing and the wall of the boring above the screen.

When installing screens, screen materials must be selected to fit both the slots on the screen and the grain size. Emplacing the gravel pack could take place during water circulation. This ensures optimum packing of the screen gravel. The gravel pack should be led at least 0.5 m above the screen to avoid bentonite choking the screen. For the same reason it is advantageous to select finer-grade screen sand for the top 0.5 m. Immediately upon completion, the soil boring must be pumped clean to achieve optimum effect.

In certain circumstances it may be necessary to add water during soil boring as well as screen installation. The addition of water may be vital to the quality of soil boring as well as of screen installation and grouting. It is a requirement that the water used in connection with drilling and screen installation is uncontaminated tap water.

The purpose of grouting the well casing is to avoid unintentional spreading of contamination as a result of the soil boring. If contaminated soil strata with clay substrata are drilled through, it is important to carry out grouting with bentonite at the clay stratum. It must always be calculated on that natural barriers (e.g. stratification of the formation) will be destroyed during soil boring. These barriers are best restored by grouting along the entire length of hole without screens. Bentonite is used for grouting, whether bentonite powder or granulate. Grouting with ”clay balls” alone will not suffice.

Granulate must always be poured into water. For this reason, water must be added to the space between casing and wall of soil boring if granulate is used above the groundwater table. An alternative is bentonite powder, which is stirred in a mixing vessel to a porridge-like consistency of bentonite slurry.

When grouting at ground level, the bentonite can be poured from ground level. When grouting at greater depth and beneath the groundwater level, liquid bentonite must be led down through a pipe (observation pipe) or pumped down using a special pump. In most cases the use of bentonite granulate is an alternative, possibly in the form of expanding bentonite. Granulate has the advantage of facilitating easier measurement of the location and thickness of the grouting during the grouting process. On the other hand, better contact between drilling hole and casing is obtained by using liquid bentonite /2/.

The location of the grouting depends on geological/hydrogeological conditions. The following figures show 4 main types of geological conditions with indications of correct location of groutings. In connection with shallow investigation soil borings (Figure 1) it is acceptable to fill the hole back up with extracted material (there should, however, always be a minimum seal of 1 m). The reason that this applies to shallow soil borings in particular is that soil boring has been carried out in the top soil strata and in groundwater aquifers near ground level. If these strata are already contaminated, the extracted material will not add any new contamination. If the top soil strata are uncontaminated, the backfill will likewise be uncontaminated.

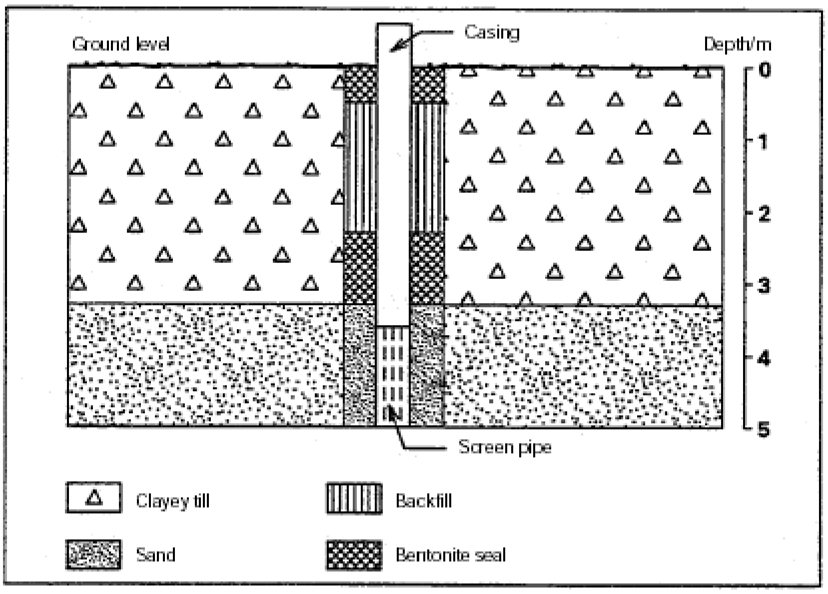


Figure 1
Minimum seal of localisation boring with screens installed in aquifer near ground level.

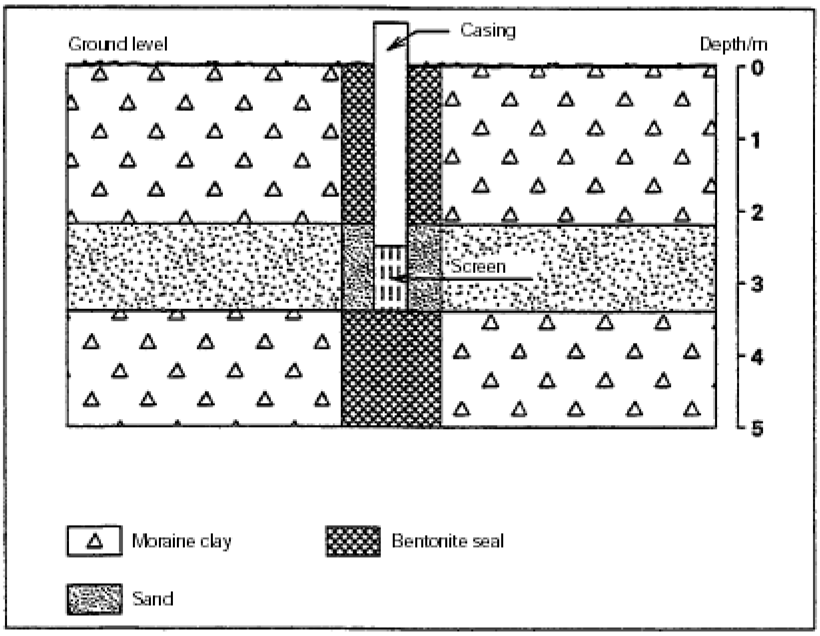


Figure 2
Grouting of deep wells with screens installed in aquifer near ground level

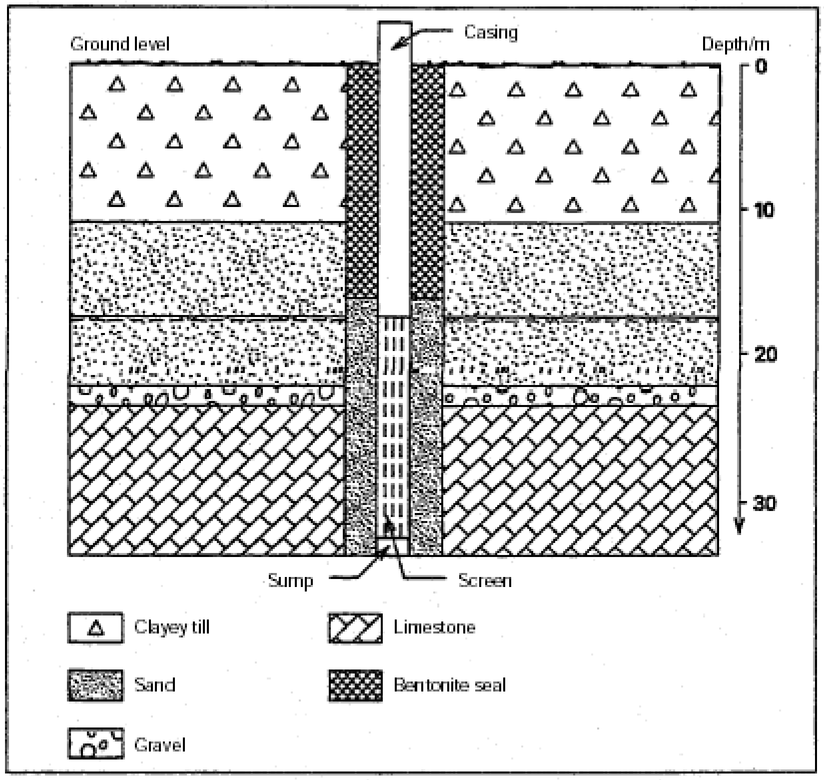


Figure 3
Grouting of deep investigation well to a deep lower aquifer (overlying gravel strata in hydraulic contact with the limestone).

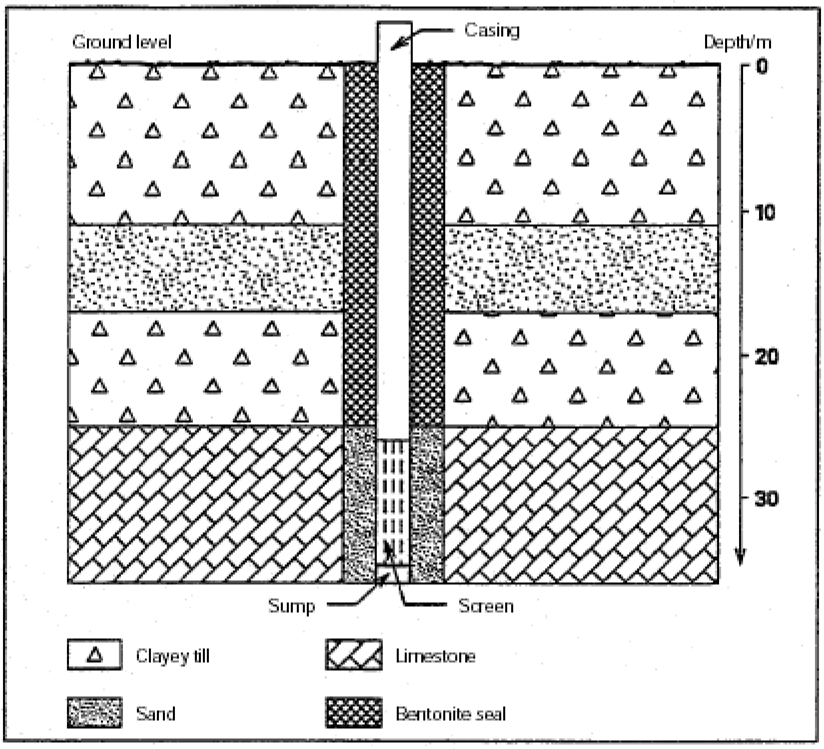


Figure 4
Grouting of deep investigation soil boring/well to deep lower aquifer (lower aquifer and aquifer near ground level are separate).

In connection with soil borings at contaminated sites it is necessary to consider the question of disposal of the extracted material. Regardless of the degree of contamination it is often unacceptable to leave the material at the investigated site.

Since disposing of contaminated soil can be costly, an aim during boring should be to separate uncontaminated soil from soil which is deemed to be contaminated. It would be advantageous to keep soil extracted from individual soil borings separate with an aim to subsequent separation of contaminated and uncontaminated soil. If there is any doubt whether or not soil is contaminated, it should be considered contaminated and treated accordingly.

Uncontaminated soil can be deposited at a landfill. However, contaminated soil must be turned in for treatment at a central processing plant. Consequently, final decisions as regards disposal must wait, pending the results of soil analyses.

All soil borings should be charted on a map and all soil borings with screens installed should also be levelled out. Borings are charted in relation to buildings or in a system of co-ordinates. In a minority of cases it may be necessary to level localisation borings in a relative datum system. In this case, a retrievable relative reference point is used. This relative fixed point is assigned the datum +180, preventing any doubt as to whether this is a relative or absolute datum. In connection with levelling the datum is set at ground level with a degree of accuracy of 0.1 metres and at observation point with a degree of accuracy of 0.01 metres (whenever possible, this should be to the highest point of the top of casing to avoid confusion). Marking the point of measurement facilitates recognition if the casing is subsequently shortened.

All borings must be safely abandoned when they are no longer needed. The guidelines in S. 15 of the Ministry of Environment and Energy Statutory Order No. 4, January 1980 must be followed when abandoning and grouting obsolete borings /3/. The owner of the boring is responsible for ensuring that abandoning procedure is carried out.

References

/1/ Hvam, T. *Markundersøgelsesmetoder* (‘Field Investigation Methods’). DGF-bulletin 5,
September 1990.

/2/ Baumann, J. *Kvalitetssoil boringer* (‘Quality Soil borings’). Geologisk Nyt 4/1996.

/3/ *Bekendtgørelse nr. 4 af 4. januar 1980 om udførelse af soil boringer efter grundvand* (‘Statutory Order No 4, 4th of January 1980 on executing soil borings for groundwater’.) The Ministry of Environment and Energy.

Soil sampling in connection with borings

For geological determination, soil samples are typically taken at 50 cm intervals; however, at least one sample must be taken for each soil stratum. When investigating the top layers of soil (0-0.5 m) it will be necessary to take samples near ground level.

It will normally be sufficient to take soil samples at 50 cm intervals as regards geological characterisation and PID measurement. Samples are often selected for chemical analysis based on this characterisation and PID measurement. Descriptions of geological characterisation is found in Appendix 4.9.

However, there are cases where it is appropriate to take samples at smaller intervals. In connection with contaminants that are strongly embedded in the soil – such as heavy metals – samples should be taken at smaller intervals around the level that is expected to be contaminated. In connection with oil contamination it may be relevant to increase sampling around groundwater level.

Soil samples are taken as normal stirred samples from a bucket auger or a solid-stem auger. To prevent cross contamination, the outer couple of centimetres must be removed using a clean trowel/scraper, whereupon the soil sample is taken using a trowel/scraper/hollow sampler and put directly into the sample container. Intact soil samples can be extracted using a hollow sampler in cases where optimum sample material is wanted under almost physically intact conditions, or where evaporation of any volatile contamination is not wanted. After purging, the sample is taken from the bottom of the soil boring using a hollow sampler as known from soil engineering /2/.

Normally, 50 g of soil is used for chemical analysis, while approximately 200-300 g is used for geological characterisation. The laboratory extracts test portions of the incoming sample in accordance with the amount needed for analysis.

There are no requirements made for packaging for soil samples that have been extracted for the purpose of geological determination. For instance, the use of plastic bags is widespread. Samples taken for chemical analysis must be packaged in a way that ensures minimum change during transport and waiting time. Soil samples for analysis for volatile substances require extraction in tight-shutting packaging. Table 1 states which types of packaging should be used. Packaging should be completely filled to avoid air above the samples. This is done in order to minimise the loss of volatile substances from the soil samples /1/. It is most practical for the laboratory carrying out the analyses to supply test tubes in connection with analyses of volatile substances.

Table 1
Packaging and storage /3/.

Substance types	Packaging	Transportation and storage	Storage life
Volatile/degradable substances Oil products + styrene, except heavy oil Chlorinated solvents Water-miscible solvents	Diaphragm jar Redcap/duran jar with Teflon lid Stainless steel tube with Teflon cover	Cool, at 4°C	Max. 24 hours
Degradable/unstable substances Phenols Mercury Chrome(VI) Cyanides	Glass with airtight lids, i.e. diaphragm or Redcap/duran jars Jam jars	Cool and dark, at 4°C	24-48 hours
Stable substances Heavy metals, Pb,Cr,Cu, i, As, Cd, Zn Phthalates Heavy oil Old tar/asphalt PAH DDT	Jam jars Nylon bags	No particular requirements, though cool and dark storage is preferable	1 month

Table 1 shows various forms of packaging, storage, and storage life for various substance classes. The samples must be delivered to the laboratory as quickly as possible. Until delivery, the samples must, if possible, be stores in a dark place at a temperature of approximately 4°C. (The samples should not be frozen, as this may cause the loss of compounds with a lower freezing point than water). The jars must be kept shut.

An extra set of samples is taken in cases where measurement using a photo ionisation detector (PID) is desired. The samples may be packed in rilsan bags, which inhibit diffusion. Guidelines for measurement using a PID detector are found in Appendix 4.7. As regards disposal of the soil samples, the same conditions apply here as for any other extracted material.

References

/1/ Jordprøvetagning på forurenede grunde, strategier, metoder og håndtering. Lossepladsprojektet. (‘Soil Sampling at Contaminated Sites; Strategies and Methods. The Landfill Project’) Report U8, April 1991.

/2/ Tage Hvam. Markundersøgelsesmetoder - mekaniske. (‘Field Investigation Methods – Mechanical’) DGF-bulletin 5, September 1990.

/3/ Vejledning om prøvetagning og analyse af jord (‘Guidelines on Soil Sampling and Analysis’). Draft for Guidelines, 1997, the Environmental Protection Agency.

Water sampling

The objective of sampling is to obtain water samples from borings which are representative of the part of the aquifer where screens have been installed as regards the parameters under scrutiny.

Water samples are usually collected from screens that have been established in the borings. As regards localisation borings, only the upper saturated zone normally has screens installed, probably using a screen with a diameter of 63 mm or 25 mm. Screen-installation depends on hydrogeological conditions and the specific objective of the boring, cf. Appendix 4.1.

A distinction should be made between well development and purging a well to removal of stagnant water. Wells should be developed immediately upon completion to ensure good performance. As regards wells, this development might be carried out using staged test pumping, cf. section 4.2.3 and /1/.

The sampling process involves three stages:

- Purging
- Sampling
- Sample storage

The groundwater of investigation wells is in contact with the air. This means, for instance, that the temperature and oxygen and carbon dioxide levels in the well and its immediate vicinity are significantly different from conditions in the aquifer. These differences may have the effect of causing differences in the content of contaminants in aquifer and well due to chemical and biological activity. In addition to this, volatile compounds may have evaporated from the water in the well. Removal of stagnant water must be carried out in the well to ensure that a water sample may be taken which is as representative of the groundwater aquifer as possible.

Either submersible pumps or suction pumps are used for removal of stagnant water. The pumps can be divided into the following types:

- Suction pumps
  - Centrifugal suction pumps
  - Peristaltic pumps (tubing pumps)
  - Vacuum pumps

- Submersible pumps
  - Centrifugal pumps (e.g. Grundfos submersible pumps)
  - Positive pressure pumps, e.g. suction pumps
  - diaphragm pumps
  - piston pumps

- gear pumps
- vane pumps

Vacuum pumps and peristaltic pumps can be used in cases where the screen pipe has a diameter of no more than 125 mm and the water table is at a maximum depth of 6-7 m below ground level. The water is usually drawn up through 10-15 mm polyethylene (PE). In connection with vacuum pumping, the tubing used for sampling should normally be used as disposable equipment to avoid any transfer of contamination between individual wells (cross contamination).

Diaphragm pumps can be used with low-yielding wells with screen with a diameter of less than 125 mm. Usually; sampling is conducted using a 10-15 mm PE or Teflon tubing. Teflon tubing are often used with diaphragm pumps and are not disposable equipment. In these cases the tubing must be thoroughly flushed with clean water when used for a different well.

Centrifugal submersible pumps are used with pipe diameters of 63 mm and more. A range of tubing can be used; from 12 mm PE (PEL) tubing with the smallest submersible pumps up to 2" tubing when using the large submersible pumps /2/.

With every well submersible pumps must be properly cleaned before use. This is best done by pumping clean water through the pump and cleaning the outside, normally also using clean water for this purpose depending on the type and extent of contamination. Depending on size, the tubing should normally be used as disposable equipment. PE tubing (PEL tubing) are recommended /2/.

High-yielding wells may have primary pumping carried out in the following way:

- In connection with primary pumping, the water should be made to pass a conductivity meter. When conductivity stabilises and becomes constant, the water is fresh aquifer water and not water from screen pipes and gravel packing (annulus water). However, quantities of water at least 10 times the water volume of screen pipes and casings must be pumped.

In low-yielding wells, removal of stagnant water may be carried out in the following way:

- If the well is pumped dry before pumping is complete, it should be pumped dry 1-4 times instead
- During pumping, pumping intake varies between the top and bottom of the water column. This particularly ensures the removal of any impurities on the water surface and at the bottom of the screen and water outside any casing is renewed.

The objective of sampling is to collect a water sample from the aquifer through the well. Sampling should be conducted immediately after removal of stagnant water. During this stage three factors are of special importance:

- Equipment must not give false positives (cross-contamination).

- Equipment must not be made of materials that adsorb or absorb substances.
- The method used must not mean that contaminant content of the sample is affected.

When vacuum and peristaltic pumps are used for water sampling, the sampling container is inserted between the well and the pump, so that the sampled water does not come into contact with the pump. In connection with pneumatic pumps and diaphragm pumps the water sample is pressed directly into the sampling flask through the tubing. In connection with submersible pumps the water is pumped directly through the pump and into the sampling flask through the tubing.

With every new consignment of sampling tubing, it is advantageous to routinely analyse a piece of the tubing in order to determine whether the tubing gives off undesired substances (and which substances, if any, are emitted).

Level specific water samples can e.g. be extracted by:

- Electrical log wells
- MPS
- In screens separated by packer and sealed casings.
- In long screens with watershed formation using several pumps, possibly controlled using heat-pulse flowmeters.

In addition to this, water samples may be extracted in connection with driven wells and cone penetration tests, etc.

When taking samples in sampling container, spraying the sample into the container must be avoided as this may occasion a considerable stripping of volatile substances from the sample.

Sorption of contaminants from the water may occur in the tubing. These effects may be minimised by shortening sample dwell time in the tubing as far as possible, and by using Teflon tubing where sorption must be completely avoided. Appendix 4.11 provides examples of forms for use in water sampling.

The packaging used for storing and during transport to the laboratory must ensure that the samples change as little as possible. Sample containers are supplied clean by the laboratory. Water samples that are to be analysed for organic parameters are kept in glass flasks with tight covers. Samples that are to be analysed for non-organic parameters such as heavy metals are often kept in plastic bottles. In connection with certain parameters of analysis the laboratory conducting the analysis will provide containers which have been specially cleaned or which contain fluids to instantly preserve samples on site.

The water samples must be stored in a cool, dark place (4°C). The lapse of time between sample taking and analysis is minimised according to analysis parameters. The samples must be delivered to the laboratory conducting the analysis on the day of sampling. Whether the sampling flasks should be completely filled and whether preservation or screening should take place in the field should be arranged with the laboratory.

References

/1/ *Bekendtgørelse nr. 4 af 4. januar 1990 om udføring af boringer efter grundvand* ('Statutory Order No. 4 of 4 January 1990 on the execution of soil borings for groundwater'). Ministry of Environment and Energy.

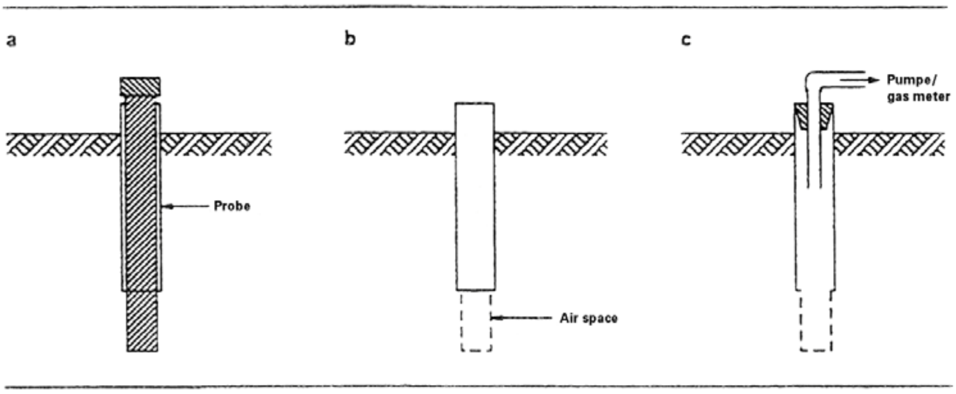
/2/ *Grundvandsprøvetagning og feltmåling* ('Groundwater Sampling and Field Testing') .
Report U3, April 1989. The Landfill Project.

Soil gas measurements

A soil gas measurement is measuring of the air in soil pores. Such measurements are carried out by pumping up a small amount of air through a hollow probe, which is either rammed or pressed into the soil. This air is then analysed.

The principle of measurement is outlined in figure 1. The driving down of the probe is carried out either manually, using e.g. a sledge hammer or a copper hammer, or mechanically, using e.g. an impact hammer, if necessary after pilot drilling. Alternatively, the probes can be pressed into the ground by hydraulic means. After being driven down, the probe is pulled slightly up. The resultant hollow space makes it possible to pump air out of the soil.

The depth of measurement is determined by the depth of the air intake, which usually corresponds to the ramming depth. Typical depths of measurement would be 0.5 – 5.0 m below ground level. With assessments of degasification to outdoor air, measurements should be made near the surface, at the top of the contamination; however, measurements should not be conducted at depths of less than 0.5 m below ground level.



a. The probe is driven to the desired depth, e.g. using a solid inner rod.
b. The probe is pulled up slightly, approximately 10 cm.
c. The measurement system is mounted at the top of the probe, allowing soil gas to be pumped up.

Figure 1
Principle for execution of soil gas measurement

When sampling in connection with assessment of indoor air, the sample should be extracted immediately below floor level.

When sampling in compact soil, the measurement represents only a very small amount of soil, preventing the measurement from indicating the presence of any contamination located lower down.

All sampling equipment that comes into direct contact with the soil gas samples must be made of a material which neither absorbs nor emits chemical substances to the soil gas samples to such an extent that it can be detected in the samples.

For probes, it is possible to use anything from black plastic water pipes to specially made pipes of stainless steel. The probe must be able to withstand being driven down and must be made of a material which can be cleaned after use or which is sufficiently cheap to allow the probe to be thrown away after use.

The tubing used should be made of Teflon or polyethylene (PEL).

The air sample is pumped up using a vacuum pump. If sampling is done at a point *after* the pump, e.g. in a tedlar bag, an oil-free diaphragm pump is used.

The pump is connected to a manometer to measure the back-pressure during pumping. If the back-pressure is heavy, leaking may occur along the sides of the probe.

Before sampling is initiated, stagnant water is removed, where an amount of air approximately 5 times the size of the air volume of the hollow space in the soil, tubing, etc. is pumped up.

The actual soil gas sample can be extracted in several way, depending among other things on the analysis parameters of the analysis method selected, e.g.:

- Extraction with a syringe through a probe in the tubing. The sample is then immediately injected into a portable gas chromatograph.
- Collection in tedlar bags, after which the samples can either be analysed in the field using a portable gas chromatograph or be brought to a laboratory for analysis. This does, however, require a short transport period, as the bags are not completely airtight.
- Collection in liquid or in adsorption tubes, e.g. charcoal sampling tubes, which are then sent to the laboratory for analysis.

During sampling , sources of error should be avoided; for instance in-leakage of outdoor air due to leaks, including leaks along the probe.

When collecting samples in liquid or tubes, the amount of air that has been collected must also be registered. For this purpose, a gas meter or a flow meter can be used, combined with records of the collection time. Using a flow meter requires a constant flow. Both gas meters and flow meters must be placed after the pump.

When taking samples in charcoal sampling tubes, tubes must remain vertical during sampling to avoid the formation of air passages in the pipes. Furthermore, if the flow (pumping) is too forceful, desorption of the most volatile components may occur. However, this is normally registered in connection with analysis, where these substances are also found in the control zone of the pipe. Collection normally takes place at a pumping capacity of 250 - 1,000 ml/min.

When collecting in charcoal sampling tubes, sampling time is somewhat longer that with other methods, but in return lower limits of detection are obtained.

Substances from probes, tubing, syringes, bags, and pumps may contaminate samples. To avoid cross contamination from probes, a sufficient supply of probes is brought along to avoid using the same probe twice during the same day. The probes are cleaned after use, using either pneumatic flushing or steam.

Syringes are cleaned through heat treatment at 150° C for at least one hour. This cleansing process is subsequently checked by injecting ‘clean air’ into the gas chromatograph using one of the syringes. Tubing and pumps are cleaned through a longer period of pumping ‘clean air’, while mildly contaminated tedlar bags can be cleaned through repeated emptying. The cleaning process for tubing, pumps, and bags is checked through subsequent collection and analysis of ‘clean air’.

Appendix 4.5

Introduction to landfill gas

This Appendix contains a brief introduction to the issue of landfill gas. For further information, see Report No. 69 from the Environmental Protection Agency /1/.

When organic waste decomposes through chemical processes, gas develops. Decomposition mainly develops methane and carbon dioxide. In addition to these, various other inorganic gases and organic vapours are developed. All the gases that develop are collectively called landfill gas. For details about processes of decomposition, see *Miljøprojekt* ('Environmental Project') No. 84 /2/. Methane production will depend on the actual conditions; e.g. decomposition in a 10 m deep landfill will be largely finished after a period of 15-30 years, given optimum conditions for degradation. Under unfavourable conditions, decomposition may last more than a 100 years.

Naturally, the rate of gas generation depends on how degradable the waste is. A landfill will typically have areas of waste with no organic content or containing largely easily decomposed waste, where gas production has already stopped. Appendix 5.1 features a simple empirical model for an estimated assessment of the rate of gas generation.

Landfill gas consists mainly of the gases methane and carbon dioxide, both of which are odourless. In addition to these, landfill gas may in some cases contain lesser amounts of other substances that have an odour.

The two main gases in landfill gas are characterised by being:

CH_4 *Methane*	CO_2 *Carbon dioxide*
· *Colourless*	· *Colourless*
· *Odourless*	· *Odourless*
· *Lighter than outdoor air*	· *Heavier than outdoor air*
· *Slightly water soluble*	· *Very water soluble*
· *Non-toxic*	· *Part of exhalation air*
· *Flammable and explosive*	· *Suffocating in high concentrations*
	· *Damaging to plant life in high concentrations*

Degradation reduces the volume of waste, causing the ground to settle. This may cause cracks and fissures in the foundations and drainage systems of any buildings on the site.

The development of gases may cause pressure above that of the atmosphere, forcing the landfill gas upwards and out of the landfill to be diluted with outdoor air. Emission will vary, among other things according to meteorological conditions, especially changes in barometric pressure. Gas transport may occur both through diffusion and through convection.

Where the area producing the gas is covered by asphalt and buildings, or where the surface is sealed off through other causes (e.g. a long period of precipitation or severe frost) the gas will be prevented from leaking up. As a consequence, pockets of gas may develop, from which gas can leak into buildings through fissures and cracks in floors and foundations, or leak sideways through natural strata of sand and gravel or through ducts and conduits.

Methane is flammable in concentrations of between 5 and 15 vol. % of outdoor air. This is called the lower and upper explosion limit, respectively. Gas mixtures with methane content above the lower explosion limit will comprise an explosion hazard when mixed with outdoor air. If methane collects in sewage systems, hollow spaces below houses, etc., there may be danger of explosion.

Carbon dioxide may constitute a hazard, as concentrations between 4 and 7 vol. % may cause unconsciousness, and higher concentrations can be lethal.

Landfill gas can also cause abnormal plant growth in gardens and open lawns; either because of an oxygen deficiency in the root zone or due to a direct toxic effect.

References

/1/ *Lossepladsgas* (‘Landfill Gas’). Report No. 69. The Environmental Protection Agency, 1993

/2/ *Alternativ lossepladsteknologi. En litteraturgennemgang* (‘Alternative landfill technology. A review of literature on the subject’). Environmental Project No. 84. The Environmental Protection Agency, 1987.

Soil gas measurement of landfill gas

Soil gas measurements are measurements of the air in soil or waste. The air is pumped up from temporary probes or from fixed measuring points. The probes are driven down manually or mechanically, if necessary after pilot drilling.

The depth of measurement is determined by the depth of the probe or the air intake of the measuring point. Typical depths of measurement are found in the interval between 0.5 – 5.0 below ground level, depending on location of the fill and the purpose of the investigation. In cases where evaporation into the outdoor air is to be examined, the measurements should be conducted near surface level, approximately between 0.5-1.0 m below ground level. When taking samples in connection with indoor air assessments for existing buildings, the samples should be taken immediately below the floor.

Unlike soil gas measurements of organic and inorganic vapours, measurement of landfill gas calls for are no special requirements regarding which material probes, tubing, and fittings are made of.

The principle for driving down a probe is illustrated in Figure 1, whereas Figure 2 shows a design proposal for a fixed measuring point.

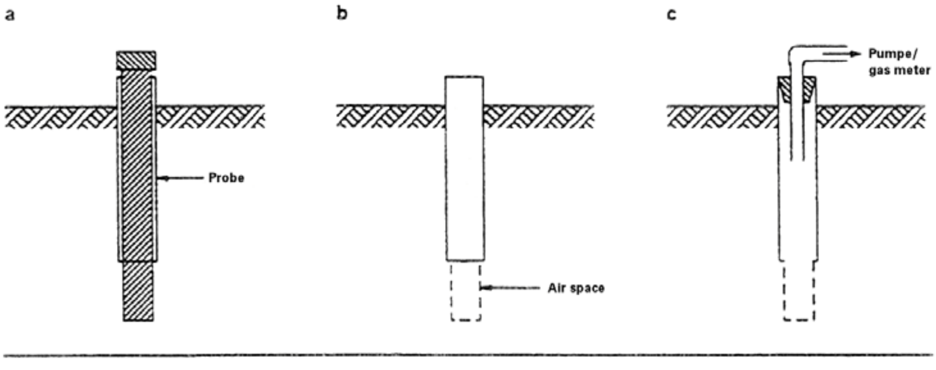


Figure 1
Principle for driving down a probe.

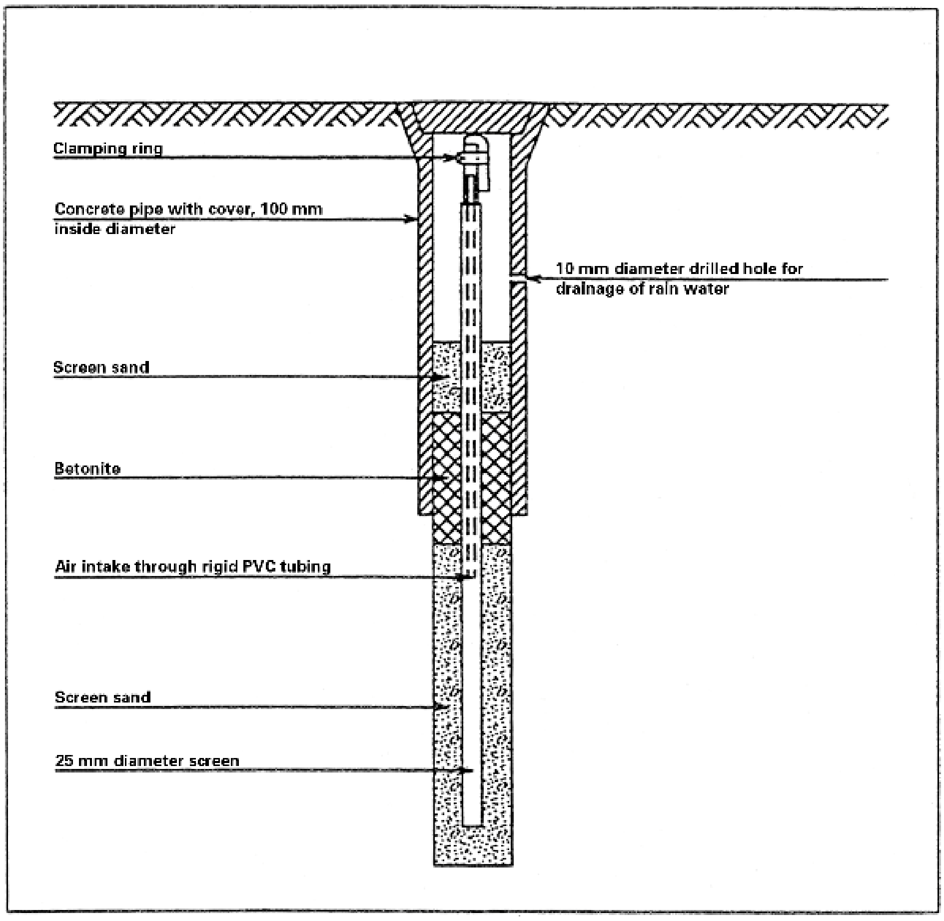


Figure 2
Design proposal for a fixed measuring point.

When measuring gas, the content of methane, carbon dioxide, and oxygen are measured using a portable metering instrument. It is recommended that methane and carbon dioxide contents are determined using infrared absorption, whereas oxygen concentrations are determined using an electrochemical cell. A description of other types of instruments can be found in Working Report No. 69 from the Environmental Protection Agency /1/.

If gas concentrations are continuously logged, continuous pumping of soil gas must be avoided, since this may disturb gas generation in the waste cell. Continuous pumping can be avoided in various ways, e.g. by setting the equipment to pump external air and soil gas at intervals. Gas measurement is recorded on a data logger when soil gas is being pumped.

If landfill gas is detected at a measuring point, soil gas pressure is measured. Such pressure measurements will indicate any pressure above that of the atmosphere caused by landfill gas contents. A liquid-charged U-pipe pressure gauge can be used for pressure measurement.

When monitoring landfill gas, gas concentrations should be seen in relation to meteorological conditions. Data might be obtained from the Danish Meteorological Institute (DMI) detailing observed barometric pressure (registered every three hours), recorded daily quantity of precipitation and recorded ground temperature from the monitoring station(s) nearest the site in question.

References

/1/ *Lossepladsgas* ('Landfill Gas'). Working report from the Environmental Protection Agency. No. 69, 1993.

Appendix 4.7

Methods of analysis /1/

	Feltmetode	Screeningsmetode	Specifik metode
TUNGMETALLER			
Arsen	EDXRF	EDXRF/ICP/	AAS-grafitovn
Bly	EDXRF	EDXRF/ICP	AAS-flamme
Cadmium	-	ICP	AAS-grafitovn
Chrom	EDXRF	EDXRF/ICP	AAS-flamme
Kobber	EDXRF	EDXRF/ICP	AAS-flamme
Kviksølv	-	ICP	AAS-coldvapour
Molybdæn		EDXRF/ICP	AAS-graft
Nikkel	EDXRF	EDXRF/ICP	AAS-graft
Zink	EDXRF	EDXRF/ICP	AAS-flamme
CYANIDER	Farve, testrør	-	SM-4500-CN syreflygtig
OLIEKOMPONENTER			
Benzen	PID/FID felt-GC Immunoassay farvereaktion testrør Fluorescensmålinger	GC/FID-pentan	GC/MS-pentan
Toluen			GC/MS-pentan
Xylen			GC/MS-pentan
Naphthalen			GC/MS-pentan
Styren			GC/MS-pentan
Benzin			-
Terpentine, min			-
Dieselolie			
KULTJÆRE			
Total PAH'er	Immunoassay	(evt. indikationer med GC/FID-pentan)	GC/MS-toluen
Benzo(a)pyren	-	-	GC/MS-toluen
ANDRE AROMATER			
Phenoler	PID/FID	(evt. indikationer med GC/FID-pentan)	GC/MS-dichlormethan*
Chlorphenoler	PID/FID		GC/MS-dichlormethan*
Pentachlorphenol	PID/FID/Immunoassay		GC/MS-dichlormethan*
Nitrophenoler	PID/FID		GC/MS-dichlormethan*
VANDOPLØSELIGE			
Acetone	Testrør-farve/PID/FID	GC/FID-vandig ekstrakt	GC/MS-vandig ekstrakt
Diethylether	PID/FID-testrør		
Isopropanol	PID/FID-testrør		
Methylisobutyl - keton	PID/FID-testrør		
DETERGENTER - ANIONISKE	-	DS 237	-
DDT	-	(evt. indikationer med GC/FID-pentan)	GC/MS-dichlormethan*
PHTHALATER **DEHP**	-	(evt. indikationer med GC/FID-pentan)	GC/MS-dichlormethan*
CHLOREREDE OPLØSNINGSMIDLER			
Vinylchlorid	PID/FID testrør felt-GC	-	GC/MS/-xylen
Chloroform		GC/FID-pentan	GC/ECD-pentan
1,1-dichlorethan		-	GC/MS/-xylen
1,2 -dichlorethan		-	GC/MS/-xylen
Dichlormethan		-	GC/MS/-xylen
Trichlorethylen		GC/FID-pentan	GC/ECD-pentan
Tetrachlormethan		GC/FID-pentan	GC/ECD-pentan
1,1,1- trichlorethan		GC/FID-pentan	GC/ECD-pentan
Tetrachlorethylen		GC/FID-pentan	GC/ECD-pentan

* Dichloromethane is undesired as an extraction agent and must be replaced.

References

/1/ *Vejledning om prøvetagning og analyse af jord* ('Guidelines on sampling and analysis of soil'). Draft for Guidelines, 1997, the Environmental Protection Agency.

Appendix 4.8

Geological assessment

During soil boring, soil samples are normally taken at intervals of 50 cm and from any strata that may occur in between these intervals. At the same time, the drilling foreman will note down his observations on the soil boring and prepare a preliminary description of the extracted soil strata and a record of stratum boundaries. The actual geological assessment of the soil samples is carried out with due reference to this information and the specific type of contamination in each individual case. This assessment sees the samples being classified on the basis of a description of soil type (lithology), geological period and formation environment. The geological assessment is supported and supplemented by information from existing information from soil borings in the area, geological literature and topographical/geological maps. The main outline of the assessment is shown in Table 1.

Table 1
Main outline of sample description. The quality of samples and the purpose of the sample description determines the degree of details and consequently the scope of the assessment..

1. Lithology	1.1 Overall designation
	1.2 Degree of hardening
	1.3 Grain size and grade
	1.4 Secondary components
	1.5 Structures
	1.6 Colour
	1.7 Mineralogy
	1.8 Carbonate content
	1.9 Common term/lithostratographical designation
2. Formation environment	
3. Period	

The geological assessment may include:

- An overall designation, written in capitals in the lithologic log, stating the dominant component of the sample, e.g. GRAVEL or SAND. Secondary component contents of the sample are written in lower-case letters in the lithologic log, e.g. clayey and/or silty, etc.
- The degree of hardening is used e.g. to separate calcareous deposits into limestone and limestone rock, where the latter is characterised by being so hard that the sample can only be scratched using a knife. In hard rock, groundwater movement will occur in fissures. It must be noted that fissures in limestone rock will only be registered during geological assessment if block or core samples of rock are available.
- The main component and secondary components, along with any fissuring, determines the permeability properties of the sample, properties influencing water movement in the strata.
- Signs or layers of fill, e.g. brick or slag etc.

- In many cases, the colour of the sample will reflect the oxidation-reduction (redox) conditions of the strata; which may influence the mobility of the contamination. In addition to this, colour/redox conditions, when seen in relation with the surrounding strata, can often provide an impression of the type and degree of contamination.
- Based on the above description and the background information, classification of the soil samples is concluded by an assessment of the geological period and formation environment of the samples. The assessments of samples are presented in lithologic logs showing the stratification of the soil borings.

Jointly, these records make it possible to correlate the geological information in a geological model which is in accordance with and related to the geological surroundings of the site.

Within the investigation site, the lithologic log provide information on geological and hydrogeological conditions as an aid for trial pumping etc. In addition to this, the records can also provide valuable information on conditions of geochemistry and dispersion with reference to contamination.

Appendix 4.9

Pumping tests

There are two normal strategies for pumping tests; step-by-step pumping test using varied yield and using constant yield. These strategies can be combined by extending the final stage as long as desired.

Before, during, and after the period of pumping, observations of the water level in a number of wells are carried out. During pumping, the discharge rate of the pump is observed. In connection with pumping tests of a duration longer than approximately one day, the barometric pressure is also measured, as are the water levels of nearby recipients, if any.

The observations will normally be conducted according to a logarithmic time scale, i.e. with short intervals in connection with start-up and after pumping is finished. Automatic data collection using pressure transducers is often seen. As a main rule, the following observation frequency should be observed for pump well and observation wells after pumping start/stop:

0-10 min	every min
10-20 min	every 2 min
20-40 min	every 5 min
40-60 min	every 10 min
60-90 min	every 15 min
90-180 min	every 30 min
180-600 min	every hour
10-24 hours	every 4 hours
1-3 days	every 6 hours
after this period	twice every 24 hours

This section concerns interpretation of pumping test data. The drawdown and recovery observed are processed as functions of time and distance and are interpreted by comparing theoretical type models, which yields values for the aquifer’s hydraulic properties; transmissivity (pump well), aquifer coefficient (observation well) and leak conditions (observation well). Other information yielded concerns the boundary conditions of the aquifer (positive boundaries (such as water courses) or negative boundaries (such as low-permeable clay barriers)). In addition to this, information on the anisotropy of the aquifer is available.

The following methods are used:

- Linear mapping with a view to assessment of water-level variations not caused by pumping (such as barometric effect).
- Single logarithmic mapping with a view to interpretation of aquifer transmissivity and aquifer coefficient.

- Double logarithmic mapping with a view to interpretation of aquifer transmissivity, aquifer coefficient and leakage.

First, the water level observations from all wells are plotted linearly using the same time scale. These plots provide information on the correlation between aquifer pressure level and e.g. variations in barometric pressure, variations in neighbouring coastal waters and/or variations in neighbouring water abstraction areas. On the basis of the linear plots, these influencing factors can be taken account of. However, it is rarely possible to carry out ideal quantitative corrections, and in reality the corrections will be semi-quantitative. In addition to this, the observation wells can be divided into three groups:

- Wells which are clearly affected by pumping
- Wells which are partly affected
- Wells which are unaffected

The theoretical method assumptions can be briefly outline as follows:

- The aquifer is homogenous and isotropic.
- The aquifer is of infinite extent.
- The radius of the pumping well is infinitesimal.
- The well is led through the entire vertical extent of the aquifer.
- Water infiltration occurs throughout the entire thickness of the aquifer.
- Water discharge from the aquifer occurs instantaneously and corresponding to drawdown.
- The hydraulic parameters are identical throughout the entire aquifer and do not vary over time.

And specifically for aquifers with leakage:

- Leakage occurs between less permeable overlying and underlying strata.
- Leakage is proportional to the drawdown in the aquifer in question.
- The hydraulic gradient in overlying and underlying strata changes instantaneously and corresponds to the drawdown in the aquifer in question.

Interpretation of pumping tests in artesian/confined aquifers with no leakage and aquifers with free water level (given the assumption that there is a small drawdown relative to the thickness of the aquifer) is based on Theis' formula:

$$h_o - h(r,t) = \frac{Q}{4\pi T} \int_u^\infty \frac{1}{y} e^{-y} dy \qquad (1)$$

$$= \frac{Q}{4\pi T} \cdot W(u) \qquad (2)$$

$$\text{hvor } u = \frac{r^2 S}{4\pi T} 1$$

$h(r,t)$ = pressure level at distance r for time t (m)
h_o = initial pressure level (m)
r = distance to pump well (m)
t = time elapsed since pump start (seconds)

Q = pump yield (m^3/sec)
T = transmissivity (m^2/sec)
S = Storativity (without unit)
$W(u)$ = well function for ‘non-leakage’ aquifers

Theis’ formula here assumes that formations above and below the aquifer are impermeable.

In connection with tensioned/artesian aquifers with leakage the following analytical equation is used:

$$h_o - h(r,t) = \frac{Q}{4\pi \cdot T} W(u, r/B) \tag{3}$$

where

r/B = leakage factor
$W(u,r/B)$ = well function for tensioned/artesian leakage aquifers (Walton function).

In connection with prolonged pumping in unconfined aquifers with delayed water discharge, interpretation must be carried out when the effect of the delayed water discharge subsides, whereupon drawdown will once again follow a Theis curve.

Two different methods are used to interpret test-pumping data. One is a single-logarithmic rectilinear adaptation method, whereas the other method is line of best fit in a double-logarithmic plot.

The single-logarithmic method is based on the fact that the integral W(u) can be represented by an infinite series. For low values of u, (1) can be approximated by:

$$h - h_o = \frac{2.3 \cdot Q}{4\pi \cdot T} \log \frac{2.25 \cdot T \cdot t}{r^2 \cdot S} \tag{4}$$

Test data with drawdown/recovery are plotted on the y-axis and time is plotted on a logarithmic x-axis. A straight line is placed through data and Δs (drawdown/recovery) is read for an interval corresponding to a factor of 10 (e.g. 10 minutes and 100 minutes). T and S are calculated in the following way:

$$T = \frac{0.183 \cdot Q}{\Delta s} \tag{5}$$

$$S = \frac{2.25 \cdot T \cdot t_o}{r^2} \tag{6}$$

where to is time (seconds) corresponding to the point of intersection between the line of best fit through the data and the line of zero drawdown/recovery.

For aquifers with no leakage, test date are plotted double logarithmically with drawdown/recovery on the y-axis and time on the x-axis. After this, a likewise double-logarithmic plot of W(u) is superimposed on u until the best possible convergence is obtained, given parallel axes. The values in pairs from drawdown/recovery data and time (s,t) and (W(u), 1/u) are called the match point; readings are taken of these. Transmissivity T and permeability k can then be calculated:

$$T = \frac{Q}{4\pi s} \quad (\text{hvis } (W(u), u) = (1,1)) \qquad (7)$$

$$k = T/m \qquad (8)$$

$$S = \frac{4 \cdot T \cdot t}{r^2} \qquad (9)$$

where

s = drawdown/recovery (m)
m = aquifer thickness (m)

Double logarithmic mapping and line of best fit are also used correspondingly in connection with aquifers with leakage. In this case, drawdown/recovery time is interpreted against Walton's leak-type curves. In this interpretation, the permeability p' (m/second) of the overlying and underlying strata can also be described by:

$$p' = \frac{(r/B)^2 \cdot T \cdot m'}{r^2} \qquad (10)$$

where m' is the thickness of the overlying and underlying strata (m).

Interpreting pumping tests requires an assessment of the extent to which actual conditions correspond to the theoretical assumptions that were described earlier. The greatest differences between actual conditions and theory are:

a) The radius of the pump well is not infinitesimal.
b) The pump well is rarely led through the entire vertical extent of the aquifer.
c) The aquifer is of finite extent.
d) The aquifer is rarely homogenous and isotropic.
e) Fissured aquifers may display double porosity and permeability may vary with pressure.
f) Leak flow is not instantaneous corresponding to decrease in water level in the aquifer.

In pre-quaternary sediments, d) and e) in particular can prevent correct determination of permeability. With a view to the outlined limitations, either corrections, a) - c), can be carried out if this is deemed relevant; or the limitations can be taken account of in connection with interpretation. In addition to this, visual assessment of bore samples compared against the geology of the site is a basic part of assessment of the hydraulic parameters of the aquifer. In this way, an acceptable estimate of transmissivity and permeability can be obtained.

In connection with interpretation, it is important to note that account often needs to be taken of corrections to the data in order to obtain useful results. In connection with tensioned/artesian aquifers it is normally necessary to correct the data to account for changes in barometric pressure (barometric effect), as this influences the water levels of wells. In addition to this, other phenomena may cause groundwater fluctuation (tidal effect, other pumping, borehole effect, partial screen installation in the aquifer, decreasing thickness, delayed water discharge, etc.). Because of this, pumping tests are open to misinterpretations, which necessitates a critical approach to using the models.

Pumping test data can be interpreted using semi-logarithmic and double-logarithmic mapping of data. Interpretations which have been carried out using double-logarithmic mapping should be considered the most accurate estimates of transmissivity, and greatest emphasis is normally given to data from the recovery. Interpretations based on single-logarithmic mapping should be considered as acceptable estimates of transmissivity.

In connection with step-by-step pumping tests, the drawdown is considered as being made up according to the following:

$$s = BQ + CQ^2$$

where B is traditionally formation loss and C is screen loss. Through analogy to pipe hydraulics it can be inferred that high C values are to be expected when the velocity (particle velocity) of the water is high, i.e. where large quantities of water pass through small cross sections. This might indicate that the well has been poorly executed. Methods of determining B and C are found in /1/. In principle, when interpretation has been carried out it is possible to calculate future drawdown in connection with any given pumping, which may assist dimensioning of any remedial equipment.

Reporting should include:

- The execution of the pumping test
- The results of the pumping test

if desired with the following supplements:

- The results of the pumping test, linear mapping
- The results of the pumping test, single-logarithmic mapping
- The results of the pumping test, t/r^2 mapping

Reference

Large amounts of useful literature concerning pumping tests are available, including guidelines from the Environmental Protection Agency /1/, which contain further references.

/1/ *Vandforsyningsplanlægning 1. Del* ('Water supply planning part 1') *Vejledning fra Miljøstyrelsen Nr. 1*, 1979. ('Guidelines from the Environmental Protection Agency No. 1, 1979')

Appendix 4.10

Standard forms

The following pages feature standard forms.

WELL LOCATION INFORMATION AND WATER-LEVEL DATA

1. Well no. ____________ Serial no. ____________
Carried out by: ____________ Year: ____________
Owner: ______________________________
Address: ______________________________
____________ Telephone: ____________

2. Sketch of well position:

1. Sketch of well construction:

WELL:

Surface level ____________m
Top of well casing level
(reference point) ____________ m
Depth to bottom,
m below reference point ____________ m
Well bottom level ____________m
Casing, diameter ____________ m
Screen diameter ____________ m
Screened interval ______m - ______ m

Accessibility to water-level measurements:
____
____

Measurement data:	Date of measurement						
	Water level, m below reference point						
	Water level						

Localised by: ______________________ Date: ____________

WATER-LEVEL DATA AND LEVELLING DATA

<table>
<tr><td colspan="9">Site: ____
Carried out by: ______________________________ Date: __________</td></tr>
<tr><td rowspan="2">Well no.</td><td rowspan="2">Reference point</td><td colspan="2">Water-level, m below reference point</td><td colspan="4">Levelling data</td><td rowspan="2">Notes</td></tr>
<tr><td>Water level</td><td>Bottom</td><td>Top of well casing (reference point)</td><td>Terrain</td><td>Bottom</td><td>Water level</td></tr>
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</table>

COLLECTION OF SOIL SAMPLES

Site: ______________________________ Date: __________ Well no. ____________
Drilling contractor: ________________ Sample collector: ____________________
Drilling method: ____
Borehole diameter: ________________ Depth to water table: _________________

Soil-type description				Odour-colour	
Strata ------- Depth	Description	Sample	Screen ------- Bottom	Description	Sample
	- --2 - --4 - --6 - --8 - --1.0 - --2 - --4 - --6 - --8 - --9 - --2.0 - --2 - --4 - --6 - --8 - --3.0 - --2 - --4 - --6 - --8 - --4.0 - --2 - --4 - --6 - --8 - --5.0 - --2 - --4 - --6 - --8 - --6.0			- --2 - --4 - --6 - --8 - --1.0 - --2 - --4 - --6 - --8 - --9 - --2.0 - --2 - --4 - --6 - --8 - --3.0 - --2 - --4 - --6 - --8 - --4.0 - --2 - --4 - --6 - --8 - --5.0 - --2 - --4 - --6 - --8 - --6.0	

COLLECTION OF WATER SAMPLES

Site: ______________________________

Borehole diameter: ______________ Screen size: ______________

Sampling method: ______________ Collected by: ______________

Well no.	Date	Well bottom, m below reference point	Water level, m below reference point	Water column, litres	Purge volume, litres	Notes

Appendix 4.10

COLLECTION OF AIR SAMPLES

Site: ______________________________

Sample collector: ______________________ Date: __________

Sample no.	Screen no.	Start time	Finish time	Final measurement	Pump no.	Flow position	Start counter	Finish counter	Pump strokes	Air per stroke, ml	Total air volume, litres

SOIL ANALYSIS – requisition form for laboratories

Site: ______________________ Sample collector: ______________________

Well no.	Depth, m below ground level	Sample no.	Date		Analysis parameters														Notes (including sample storage until submission)
			Sample collection	Sample submission															

WATER ANALYSIS– requisition form for laboratories

Site: ____________________ Sample collector: ____________________

Well no.	Depth, m below ground level	Sample no.	Date		Analysis parameters														Notes (including sample storage until submission)
			Sample collection	Sample submission															

Gas production - an empirical model for approximate estimates

This Appendix describes a model for assessing the rate of gas production as a function of time. Many different models have been developed, none of which are capable of taking into account all the factors which influence gas generation. Since knowledge of the specific parameters is also often inadequate, using a simple 1st order degradation model is suggested. For this reason, the main use of the model is to provide estimates for consideration.

Calculation formulae

The rate of gas production is given by /1/:

$$\frac{dP}{dt} = P_{tot} \cdot k \cdot e^{-kt} \qquad (1)$$

where: dP/dt = annual gas production ($Nm^3/t \cdot year$)
P_{tot} = total gas production (Nm^3/t)
k = the degradation constant ($year^{-1}$)
t = time (years)

The degradation constant is given by:

$$k = \frac{\ln 2}{t_{½}} \qquad (2)$$

where: $t_{½}$ = half-life period (years)

Data basis

The calculation is based on data concerning:

- Total gas production
- Half-life period

Literature on the subject states that, given optimum conditions, total gas production lies between 320-430 Nm^3 per ton of waste. However, in actual fact total gas production will be considerable smaller. Values of *180-210* Nm^3/t *for total gas production* have been measured in Danish landfills. These values are supported by Swedish and German calculations.

Empirical figures for half-life periods are listed in Table 1.

Table 1
Half-life periods for types of waste.

Half-life period, $t_{½}$	Years
Domestic waste /2/	1
Sludge /2/	2
Industrial and commercial waste /2/	5
Bulky waste /2/	10
Building and construction waste /2/	15
Easily decomposable waste /3/	1-1.5
Moderately degradable waste /3/	15-25

If the investigation in question has included measurements of simultaneous values of water content and waste temperature, half-life periods can be determined as a function of this /1/.

Example

On the basis of the calculation formulae (equation 1 and 2), curves of annual gas generation and remaining gas potential can be traced. Figure 1 shows gas generation and remaining gas potential as functions of time after disposal for a waste cell with a total gas generation (P_{tot}) of 200 Nm^3/t and a half-life period $t_{½}$ of 10 years.

The annual gas generation dP/dt after 10 years is calculated as follows:

$$k = \frac{\ln 2}{t½} = \frac{\ln 2}{10} = 0.07$$

$$\begin{aligned} dP/dt &= P_{tot} \cdot k \cdot e^{-k \cdot t} \\ &= 200 \cdot 0.07 \cdot e^{-0.07 \cdot 10} \\ &= 6.95\ Nm^3/t \end{aligned}$$

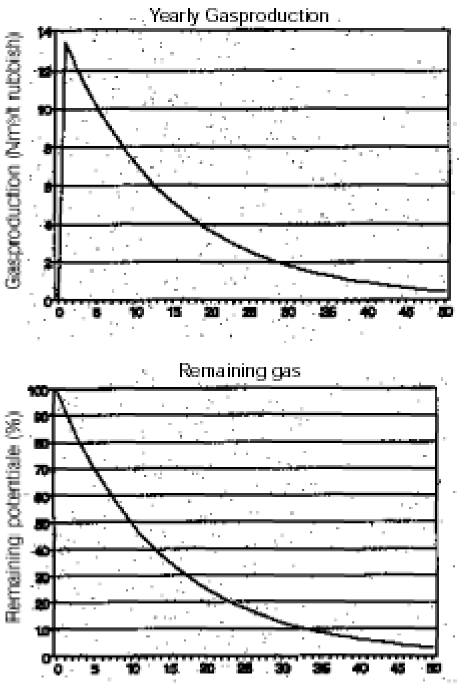

Figure 1.
Yealy gasproduction and remaining gas as a function of time after deposition in a landfill with halftime = 10 years

References

/1/ *Lossepladsgas – Transport og Produktion* ('Landfill Gas – Transport and Generation')
Erling Vincentz Fisher
Examination project, spring 1992, the Technical University of Denmark.

/2/ *Gas i lossepladser* ('Gas in Landfills')
ATV meeting, March 1993.

/3/ *Noter om: Kontrollerede lossepladser* ('Notes on Controlled Landfills')
Thomas H. Christensen and others.
Teknisk forlag, 1982.

Appendix 5.2

Convection model for gas invasion into surrounding buildings

This Appendix describes a model for landfill-gas transport into buildings placed outside of landfill area producing gas. The model, which calculates the convection contribution, is based on a worst-case scenario, where the entire quantity of gas being transported enters the building /1/. As such, this convection model is simpler than the model for convection contribution which is described in Appendix 5.3.

Calculation assumptions and formulae

A landfill with soil gas concentration C_p is situated at a distance x from a building, as shown in Figure 1. Gas transport is assumed to take place solely in the upper unsaturated zone. The landfill has a positive pressure P_s in relation to pressure at the building. Atmospheric pressure is P_{atm}.

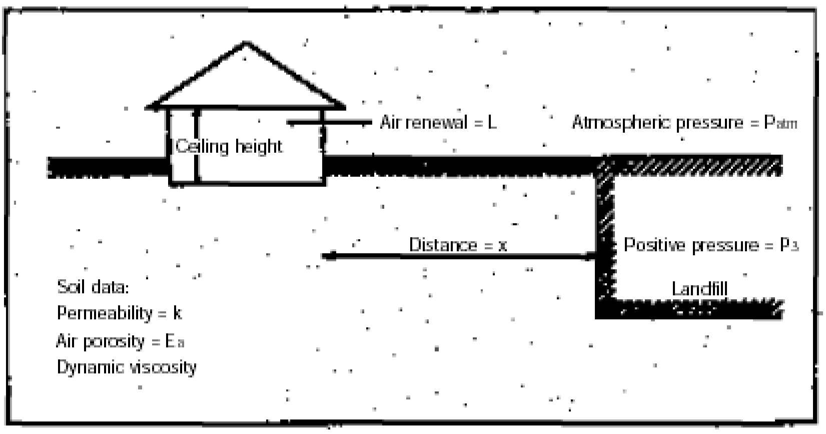

Figure 1
Diagrammatic presentation of model for transport of landfill gas to surrounding buildings.

In this model, the extent of the landfill is assumed to be much greater than the distance to the building. The following applies to the situation of equilibrium:

$$\alpha = \frac{C_i}{C_p} = \frac{k \cdot P_s}{\mu \cdot x \cdot l \cdot L} \qquad (1)$$

where:

α = the ratio between inside-air concentration of the building and soil gas concentration of the landfill
k = air permeability (m^2)
P_s = landfill positive pressure (Pa)
μ = dynamic viscosity (kg/m · s)
x = distance (m)
l = ceiling height (m)
L = air renewal (s^{-1})

The period elapsing before establishment of pressure gradient τ_{ssp} (Equation 2) and equilibrium concentration τ_{ssc} (Equation 3) is given by

$$\tau_{ssp} = \frac{\mu \cdot E_a \cdot x^2}{k \cdot P_{atm}} \qquad (2)$$

$$\tau_{ssc} = \frac{\mu \cdot E_a \cdot x^2}{k \cdot P_s} \qquad (3)$$

where: E_a = air porosity (unitless)
P_{atm} = atmospheric pressure (Pa)

Data basis

The calculation formulae (equations 1-3) state the necessary data. Empirical data for input data are listed in Table 1. For constructional data, see Table 1 in Appendix 5.3.

Table 1
Empirical data for transport model data.

Air permeability, k	m^2
• Clayey till [1)] • Fine sand • Coarse sand	10^{-13} 10^{-12} $2 \cdot 10^{-11}$
Positive pressure in the site, P_s	Pa
• Typical values measured	0 – 2,000
Dynamic viscosity, μ	kg/m·s
• Air • Methane • Carbon dioxide	$1.8 \cdot 10^{-5}$ $1.1 \cdot 10^{-5}$ $1.5 \cdot 10^{-5}$
Air porosity, E_a	unitless
• Loam • Sandy loam • Clay • Sand	0.1 0.1 0.1 0.3

[1)] Estimated value. See also Table 2 and 3 in Appendix 5.8.

References

/1/ Little, J.C., Daisey, J.M. and Nazaroff, W.W:
Transport of Subsurface Contaminants into Buildings.
An Exposure Pathway for Volatile Organics.
Environmental Science and Technology.
Vol. 26. No. 11, 1992, p. 2,058 – 2,066.

Evaporation of volatile substances from soil

1. Background

For soil or groundwater contamination by volatile substances, these substances might evaporate, causing danger of inhalation of harmful fumes both in open areas and indoors.

Evaporation from soil will often be one out of several contributors outside as well as indoors. Other contributions to air contamination will come from traffic, neighbouring enterprises, gas emission from construction materials etc. as well as from smoking, leisure activities/hobbies and so forth.

This Appendix provides a description of the principles of calculating the resultant contribution to indoor and outdoor air concentrations from volatile organic contaminants in soil.

Data regarding the following issues are used as the basis for calculations:

- contamination type and concentration, including physico-chemical data for the specific contamination components
- depth of contamination and its relative position to the groundwater table
- soil strata condition
- building parameters
- temperature and pressure conditions
- wind velocities etc.

Building parameters include the following data:

- constructions on soil. Wooden floors or concrete floors. If floors are concrete, environmental classification, reinforcements conditions, the age of the floor etc. must be determined.
- ventilation conditions of the building.
- ceiling heights of rooms.

Models for calculation of release and transport of vaporous components in soil are a relatively new occurrence, and not all formulae are equally well supported by experiments. In Denmark as well as elsewhere, work is continuously being done to acquire better knowledge of the mechanisms governing evaporation of volatile substances from soil, but as yet, models for calculating contributions from contaminated soil to indoor air and outdoor air are subject to a relatively high degree of uncertainty.

The fundamental model of calculation described here comprises three parts:

- Phase distribution in soil, i.e. the distribution of individual contamination components between soil gas, soil water, a phase adsorbed to the soil matrix and possibly NAPL contamination. Using this data, soil gas concentrations of contaminants can be calculated on the basis of total soil contaminant concentrations.
- Diffusion up through the soil and floor constructions: Here, evaporation through the soil is calculated on the basis of contaminant concentrations in the soil gas of the contaminated zone and the relative depth to this zone. Diffusion contributions to indoor air and outdoor air are subsequently calculated on the basis of this evaporation.
- Convection through floor constructions: Convection contribution to indoor air is calculated on the basis on the previously calculated diffusive transport to the undersurface of the building.

2 Calculation principles and assumptions

2.1 Phase distribution

In cases where contamination is situated in the unsaturated zone, where consequently the soil contains soil particles, soil water and soil gas, soil gas concentrations can be calculated when the phase distribution of the substance between these three phases – and any contamination with a NAPL - is known.

This calculation could be carried out using a model which among other places is described in Mackay & Paterson, 1981 /5/.

In its simplest form, the model assumes equilibrium of contamination-component content in the different phases and constant conditions. In addition to this, the model assumes that contamination components in their vaporous phases behave as ideal gases, and that components in their aqueous phases behave as ideal diluted solutions. Other versions of the model exist capable of taking into account that equilibrium has not occurred, and that decomposition and/or transport of substances is occurring. In the following, the simplest version of the model is the basis of the description.

The principle of the calculation is that the soil's maximum capacity for contamination components is calculated on the basis of saturation concentrations in air and water and the distribution ratio between water and soil particles. It is then assumed that the distribution ratio between the three phases will be the same as they would be in case of saturation, even given other concentrations, and that equilibrium has been established. Soil gas concentration at a given total concentration can be calculated on this basis.

If contamination occurs in the groundwater, soil gas concentrations just above the groundwater table can be determined – again, assuming equilibrium between the two phases – on the basis of the relationship between the contamination component’s partial pressure in air and solubility in water (Henry's law). It must be noted that diffusion in the aqueous phase is several orders less than diffusion in the vaporous phase, and that for this reason, aqueous-phase diffusion could limit evaporation from the groundwater to soil gas in the unsaturated zone. No account is taken of this in the calculations.

For contamination caused by products consisting of several components (such as oil and tar), individual components may interact, for instance by dissolving in each other or possibly by reacting chemically with each other. This influences vapour pressure as well as solubility and adsorption of individual components. However, for mixtures made up of neutral components (for instance oil) this deviation will be minor in relation to the other uncertainties in the calculations.

If organic acids are part of the mixture (such as for instance phenol in tar), the deviation will be significant, especially as regards the organic acid (for phenol in tar the resultant vapour concentration will typically be smaller by a factor of 10 due to the mixture). In principle, account can be taken of this by including activity coefficients in the equilibrium expression for individual components in the relevant mixture.

In connection with calculations for mixtures, account must always be taken of the ratio of mixture by involving the molar fraction for each individual component and during each separate stage of calculation.

2.2 Gaseous transport through soil

The model for evaporation described here is solely concerned with diffusive transport in the soil gas in soil. In connection with soil contamination, gas generation resulting in positive pressure is rare except, for example, in cases involving contamination components placed at an actual landfill. For this reason, convective transport through soil strata can be discounted in such cases.

Substance-specific parameters aside, the diffusion of a contamination component will depend on soil type and soil porosity and water content. In addition to this, the soil may comprise several strata with different properties.

Variations in barometric pressure may cause a certain pumping effect, consequently causing a flow of air in the topmost soil strata. Similarly, ventilation in buildings situated near the contaminated soil can also cause convective air flow in the soil, air flow which influences vapour transport; as will significant differences in temperature.

In connection with a contamination component’s diffusion through soil strata, double solution in seeping rainwater and sorption onto soil particles could occur. In constant conditions and on average, double solution in rainwater for soil strata up to a thickness of a couple of metres will be 1-2 orders less than the diffusive rising flux, and consequently will be of no significance. With larger amounts of water seeping over a shorter period of time, double solution may be significant.

Among other things, sorption on soil particles will depend on the water content of the soil, since sorption in damp soil will be up to two orders less than in dry soil.

Biological degradation, especially during summer, can probably cause a reduction of vapour concentration in certain contamination components in the uppermost soil strata, and hence a reduction of evaporation.

Conversely, the partial pressure of most contamination components will display considerable temperature dependence, causing an increase of evaporation during period of summer.

Finally, during hot summers and in green and cultivated areas, upward water transport in the soil due to evapotranspiration may occur. In principle, this can promote substance transport to the surface.

The simple model described here does not include convective transport through soil strata caused by any differences in pressure, nor does it address diffusion in soil water, double solution/sorption of vapours, degradation, temperature dependence, or evapotranspiration.

2.3 Diffusive contribution to outdoor air concentration

The diffusive contribution to concentration in outdoor air can be calculated based on the flux to the ground surface on the basis of e.g. the following assumption:

- Mixing of the evaporated contamination component occurs in the air within a certain height above ground level. This height depends on wind velocity and on the location of the desired point of concentration calculations. It is assumed here that the greatest concentrations are found downwind at the end of the contaminated site.

2.4 Diffusive contributions to indoor air concentrations

The diffusive contribution to indoor air concentrations can be calculated on the basis of the flux to the undersurface of buildings based on the following assumption:

- diffusion of contamination components occurs to the undersurface of buildings and further upwards through floor constructions,

- the diffusive flux of contamination components are mixed with the indoor atmospheric environment in the room closest to the ground.

2.5 Convective contributions to indoor air concentrations

The convective contribution to indoor air concentrations can be calculated on the basis of calculated diffusive to the undersurface of buildings under the following assumptions:

- calculations are made for only one type of floor, concrete deck,

- crack spacing and crack width can be estimated on the basis of the reinforcement conditions, thickness, and environmental classification of the concrete,
- the convective contribution to indoor air can be calculated on the basis of the estimated crack spacing and crack width in conjunction with building vacuum,
- the convective flux of contamination components through the concrete deck is mixed with the indoor atmospheric environment in the room closest to the ground.

3 Calculation formulae

In the following, the formulae used in the calculation model are described in their simplest form, i.e. given all the simplifying assumptions that have been mentioned earlier. The objective is to provide a description of the order and principles of the calculations. In specific cases, careful consideration should be given to which simplifying assumptions can be made.

3.1 Phase distribution in soil

The total volume of soil can be seen as the sum of the soil phases, see equation 1.

Equation 1

$$V_L + V_V + V_J = 1$$

where:

V_L = relative volumetric proportion of air in soil
V_V = relative volumetric proportion of water in soil
V_J = relative volumetric proportion of soil particles in soil.

Here, $V_L + V_V$ equals total porosity.

Maximum contaminant content of one cubic metre (1 m^3)of soil distributed across the three phases can be calculated as follows, see Equations 2-10:

Equation 2

In the air phase of soil (soil gas):

$$M_{L,\,max} = V_L \cdot C_{L,\,max} \text{ (mg/m}^3\text{)}$$

where: $M_{L,max}$ = maximum contaminant amount in soil gas (mg/m^3 soil volume)
$C_{L,max}$ = saturated vapour concentration of contaminant (mg/m^3 soil gas).

$C_{L,max}$ can be calculated on the basis of the contaminant's partial pressure by means of the law of ideal gases:

Equation 3

$$C_{L,max} = \frac{p \cdot m \cdot 10^3}{R \cdot T} \text{ (mg/ m}^3\text{)}$$

where: p =contaminant partial pressure (N/m^2)
m =contaminant molecular weight (g/mol)
R =the gas constant ($J/mol \cdot K$)
T = temperature in K.

T = 298 K (25°C) is used as standard.

In the water phase of soil (soil water):

Equation 4

$M_{V,max} = V_V \cdot S$ (mg/m^3)

where: $M_{V,max}$ = maximum contaminant amount in soil water (mg/m^3 soil volume)
S = water solubility of contaminant (mg/m^3 soil water).

At equilibrium, the relation between contaminant partial pressure and water-phase concentration equals constant of Henry's law, H:

Equation 5

$$H = \frac{p}{S} \ (N \cdot m/mg) \text{ eller } S = \frac{p}{H} \ (mg/m^3)$$

The use of Henry's law assumes that in vapour form, the contaminant behaves like an ideal gas, and that in solution form, it behaves like an ideal diluted solution. Partial pressure and solubility must be registered at the same temperatures.

Equation 6

In the particle phase of soil:

$M_{J,max} = V_J \cdot d \cdot J_{max} \cdot 10^3$ (mg/m^3)

Where:

$M_{J,max}$ = maximum contaminant amount which has adsorbed to the organic fraction of soil particles (mg/m^3 soil volume)
d = soil particle density (kg/l)
J_{max} = amount of contaminant which adsorbs to the organic fraction of soil particles at equilibrium with saturated solution in water (mg/kg)

J_{max} can be represented by S as follows:

Equation 7

$J_{max} = K_D \cdot S \cdot 10^{-3}$ (mg/kg)

Where: K_D= contaminant ratio soil/water (l/kg).

Given the assumption that adsorption occurs only onto the soil's organic-substance content, which can be a reasonable approximation as regards organic contaminants, K_D can be calculated based on the contaminant's ratio between organic carbon and water, and the soil's organic carbon content.

Equation 8

$K_D = K_{oc} \cdot f_{oc}$ (l/kg)

Where: K_{oc}= contaminant ratio between organic carbon and water (1/kg)
f_{oc} =soil content of organic carbon (relative weight proportion).

Consequently, $M_{J,max}$ can be written:

Equation 9
$M_{J,max} = V_J \cdot d \cdot K_{oc} \cdot f_{oc} \cdot S$ (mg/m^3)

For most types of contaminants, the contaminant’s ratio between organic carbon and water can be estimated based on the octanol/water ratio $K_{ow.}$, “most types” indicating aromatics, PAHs, aliphatic hydrocarbons and chlorinated solvents, cf. Equation 10a. In connection with phenols, which are weak organic acids, pH is also included in the estimate, Equation 10b /6/.

Equation 10a
The following formula is used for contaminants with log K_{ow} less than 5 and soil with f_{oc} greater than 0.1 %:

$$\log K_{oc} = 1.04 \cdot \log K_{ow} - 0.84$$

In connection with organic acids (for example chlorophenols), K_D can be calculated on the basis of:

Equation 10b
$$K_D = f_{oc} \cdot k_{ow}^{0,82} \cdot (1.05 \cdot \Phi_n + 0.026 \cdot (1-\Phi_n))$$

Where: Φ_n = neutral acid fraction (unitless).

The neutral acid fraction can be calculated on the basis of:

$$\Phi_n = \frac{1}{1+10^{pH-pK_a}}$$

where pK_a = contaminant acid dissociation constant.

The formula applies if:

$$pH - pK_a < 1.5 \text{ and } f_{oc} > 0.001$$

It follows that the maximum soil capacity for contaminants (immediately preceding the occurrence of NAPL) will be:

$$M_{L,max} + M_{V,max} + M_{J,max}$$

The distribution of the contaminant onto the three soil phases can be calculated on the basis of the above-mentioned assumption that the relative proportion of the three phases in soil is independent of total soil concentration.

The following applies to the air phase of soil:

Equation 11

$$f_L = \frac{M_{L,max}}{M_{L,max} + M_{V,max} + M_{J,max}} = \frac{M_L}{M_L + M_V + M_J}$$

where:

f_L =relative contaminant ratio in soil gas in proportion to total content (unitless).

M_L,M_V,M= actual/relevant contaminant amount in each of the three phases (mg/m^3 soil).

On the basis of Equations 2, 4 and 9 and Equation 3 and 5, f_L can be rewritten as follows:

Equation 12

$$f_L = \frac{V_L \cdot C_{L,max}}{V_L \cdot C_{L,max} + V_V \cdot S + V_J \cdot d \cdot K_{oc} \cdot f_{oc} \cdot S}$$

$$= \frac{V_L \cdot \frac{p \cdot m \cdot 10^3}{R \cdot T}}{V_L \cdot \frac{p \cdot m \cdot 10^3}{R \cdot T} + V_V \cdot \frac{p}{H} + V_J \cdot \frac{d \cdot K_{oc} \cdot f_{oc} \cdot p}{H}}$$

$$= \frac{V_L \cdot \frac{m \cdot 10^3}{R \cdot T}}{V_L \cdot \frac{m \cdot 10^3}{R \cdot T} + V_V \cdot \frac{1}{H} + V_J \cdot \frac{d \cdot K_{oc} \cdot f_{oc}}{H}}$$

On the basis of a given total concentration in soil C_T (mg/kg soil volume) the contaminant volume in the M_L can then be established:

Equation 13

$M_L = f_L \cdot C_T \cdot \rho \cdot 10^3$ (mg/m^3 soil volume)

where: ρ = soil density (kg/l)

Contaminant concentration in soil gas, C_L, is then calculated on the basis of contaminant concentration in soil, C_T, see Equation 14.

Equation 14

$$C_L = \frac{M_L}{V_L} = \frac{f_L \cdot C_T \cdot \rho \cdot 10^3}{V_L} \text{ (mg/ m}^3\text{ poreluft)}$$

Remember that C_L cannot exceed $C_{L,max}$.

If the soil has a NAPL, i.e. saturated conditions, it is easiest to calculate $C_L = C_{L,max}$ directly on the basis of the partial pressure, see Equation 3.

3.2 Diffusion up through soil

Given a state of equilibrium, gas transport from contaminated soil to the ground surface by means of diffusion in soil gas can be described by means of Fick's law of diffusion:

Equation 15

$$J = \div N \cdot D_L \frac{C_o - C_L}{X}$$

where: J = flux (evaporation) (mg/(m^2 · s))
N = material constant (unitless)
D_L = contaminant diffusion coefficient in air (m^2/s)
X = depth corresponding to concentration C_L (m)
C_o = background concentration at the site (mg/m^3), can be set at 0, if it is significantly less than C_L.

Equation 16

In connection with a number of substances, the diffusion coefficient in air can be found in Lugg /9/ or be estimated on the basis of the following formula /6/:

$$D_2 = D_1 \sqrt{\frac{m_1}{m_2}}$$

where:

D_1 and D_2 = diffusion coefficients in air of contaminants (m^2/s)

m_1 and m_2 = contaminant molecular weights (g/mol)

If gas-phase transport occurs through various different soil strata the flux will be:

Equation 17

$$J = \frac{-(N_1 \cdot N_2 \ldots . N_n) \cdot D_L \cdot (C_o - C_L)}{N_2 \cdot N_3 \ldots N_n \cdot X_1 + N_1 \cdot N_3 \ldots N_n \cdot X_2 + N_1 \ldots N_{u-1} \cdot X_n}$$

where: N_1 to N_n are the material constants for the various strata and X_1 to X_n each represent the thickness of each individual soil-strata.

One possible expression for N for soil would be, for example, Millington /8/.

Equation 18

$N = (V_L^{3,33} / (V_L + V_v,)^2)$

where $V_L + V_V$ is total porosity.

By substituting Equations 14 and 17 into Equation 15 and by assuming that C_o can be set at 0, an expression for the flux up through the soil as a function of soil concentration is obtained:

Equation 19

$$J = V_L^{3,33} \cdot D_L \cdot f_L \cdot C_T \cdot \rho \cdot 10^3 / (X \cdot (V_L + V_V)^2 \cdot V_L)$$
$$= V_L^{2,33} \cdot D_L \cdot f_L \cdot C_T \cdot \rho \cdot 10^3 / (X \cdot (V_L + V_V)^2)$$

3.3 Diffusive contribution to concentrations in outdoor air

The mixing of the flux J in outdoor air will result in a contribution to outdoor air concentration, C_u (mg/m^3).

Given the assumption that the highest value for C_u is found downwind, at the end of the contaminated area, C_u can be calculated on the basis of the following assumption:

The mass-flow velocity, Q_1, perpendicular up through the contaminated area equals the mass-flow velocity, Q_2, through a vertical section perpendicular to the wind direction at the end of the contaminated area, given that diffusion in the air over the contaminated area is disregarded. This assumption is not unreasonable in connection with the relatively short distances involved here.

This gives the following:

Equation 20

$$Q_1 = Q_2 \text{ (mg/s)} \Rightarrow$$
$$A_1 \cdot J = A_2 \cdot v \cdot C_u \Rightarrow$$
$$l \cdot b \cdot J = b \cdot h \cdot v \cdot C_u$$

or

$$C_u = \frac{J \cdot l}{v \cdot h} \text{ (mg/ m}^3\text{)}$$

where: C_u = diffusive contamination contributions to outdoor air (mg/m^3)
Q_1 = mass-flow velocity up through the contaminated area (mg/s)
Q_2 = mass-flow velocity through a vertical section perpendicular to the wind direction (mg/s)
v = wind velocity (m/s)
A_1 = area of contaminated area (m^2)
A_2 = area of vertical section perpendicular to the wind direction (m^2)
l = length of the contaminated area in the wind direction (m)
b = width of the contaminated area perpendicular to wind direction (m)
h = mixing height in outdoor air at the downwind end of the contaminated area (m).

In order for C_u to be comparable to the evaporation criteria for a substance, wind velocity, v, must in principle equal the velocity which can be measured at the point in question indicated as the 1% fractile.

Based on experience, at such low wind velocities v (≤ 2 m/s) the mixing height, h, can be set at 0.08 times the length of the contaminated area, 1. If this is substituted into the Equation 20 the result is:

Equation 21

$$C_u = \frac{J \cdot 1}{v \cdot 0.08 \cdot 1} = \frac{J}{v \cdot 0.08} \text{ (m/s)}$$

(J is determined by means of Equation 19).

or

Equation 22

$$J = C_u \cdot v \cdot 0{,}08 \text{ (mg/m}^2 \cdot \text{s)}$$

A wind velocity of 0.1 m/s is used, which corresponds to calm conditions. However, a wind velocity of 1 m/s is used for substances where the criteria for acceptance are determined on the basis of long-term effects; this includes carcinogens.

3.4 Diffusive contribution to indoor air contamination concentration

The mixing of the flux J in the indoor air will result in an indoor air contribution, C_i (mg/m^3).

The mass-flow velocity, Q_1, perpendicular up through the contaminated area into the building equals the mass-flow velocity, , Q_2, generated through air renewal in the building. This gives the following:

Equation 23

$$Q_1 = Q_2 \text{ (mg/s)} \Rightarrow$$
$$A_1 \cdot J = A_2 \cdot L_h \cdot L_s \cdot C_i$$

Given the conservative assumption that $A_1 = A_2$ (contamination occurs under the entire room), the following is arrived at:

Equation 24

$$C_i = \frac{J}{L_h \cdot L_s} \text{ (mg/ m}^3\text{)}$$

where: C_i = diffusive contamination contribution to indoor air (mg/m^3)
Q_1 = mass-flow velocity up through the contaminated area (mg/s)
Q_2 = mass-flow velocity generated through air renewal (mg/s)
A_1 = contaminated area (m^2)
A_2 = building area (m^2)
L_h = ceiling height in building (m)
L_s = air renewal in building (s^{-1})

3.5 Convective and diffusive contribution through reinforced concrete deck to indoor air contamination concentration

The convection of contaminants can occur through cracks in concrete deck and through leaking joints and leaks in pipe lead-ins etc. Calculations regarding shrinkage cracks due to drying shrinkage can be carried out by means of formulae and directions which can be found in the Danish standard specifications for concrete constructions DS411 /3/ and in *Beton-Bogen* (‘The Concrete Book’) /4/.

The calculation of air transport through cracks have been carried out according to Baker, Sharples & Ward /7/.

3.5.1 Calculating crack length and width

The crack parameter a_w can be calculated based on the following:

Equation 25

$$a_w = \frac{A_{cef}}{\Sigma d_w}$$

where a_w = crack parameter (mm)
A_{cef} = active concrete area subject to tensile force (mm^2)
d_w = crack-determining diameters of reinforcement bars (mm)

The active concrete area subject to tensile force A_{cef} can be calculated as:

Equation 26

$$A_{cef} = h_b \cdot b_b$$

where h_b = thickness of concrete deck (mm)
b_b = observed concrete-deck width (mm)

Equation 27

$$d_w = k \cdot d_a$$

where d_a = nominal diameter of reinforcement (mm)
k = constant which depends on the type of reinforcement, cf. the table below

Reinforcement	k
Ribbed bars & tentor steel	1.0
Plain reinforcement	0.5

Consequently the crack parameter will be

Equation 28

$$a_w = h_b \cdot \frac{b_b}{(b_b - \Delta b) / \Delta b \cdot d_w} = \frac{h_b \cdot \Delta b}{(1 - \Delta b / b_b) \cdot d_w}$$

where Δb = the distance between reinforcement bars (mm).

For $b >> \Delta b$ we have

Equation 29

$$a_w = \frac{h_b \cdot \Delta b}{d_w}$$

The free shrinkage strain ε_s can be estimated on the basis of the following empirical formula for the mean value

Equation 30

$$\varepsilon_s = \varepsilon_c \cdot k_b \cdot k_d \cdot k_t$$

where

ε_s = shrinkage strain (%)

ε_c = base shrinkage (%)

k_b = coefficient taking into account influences from the composition of concrete (unitless)

k_d = coefficient taking into account dimension (unitless)

k_t = coefficient taking into account shrinkage time (unitless)

Base shrinkage can be calculated as

Equation 31

Base shrinkage can be calculated as

$$\varepsilon_c = \frac{0.089 \cdot (100 - RF)}{167 - RF}$$

where RF = relative air humidity (%)

k_b is calculated based on the composition of concrete

Equation 32

$k_b = 0.007 \cdot CM \cdot (v/c + 0.333) \cdot v/c$

where CM = cement content (kg/m^3)

v/c = water/cement ratio (unitless)

The equivalent radius r and k_d are calculated by means of the following formulae:

Equation 33

$$r = \frac{2 \cdot b_b \cdot h_b}{b_b} = 2 \cdot h_b$$

where r = equivalent radius of construction (mm)

Equation 34

$$k_d = \frac{0.25 \cdot (852 + r)}{132 + r}$$

The influence of time:

Equation 35

$$k_t = \frac{t_s^{\alpha}}{t_s^{\alpha} + t_0}$$

where t_s = shrinkage time (days)

t_o, α and β are auxiliary parameters (unitless)

Equation 36

$$t_0 = 9 \cdot (\sqrt{10})^{\alpha \cdot \beta}$$

$$\alpha = 0.75 + 0.125 \cdot \beta$$

$$\beta = \frac{\ln(0.02 \cdot r)}{\ln 2}$$

3.5.2 Calculating reinforcement tension

The calculation is carried out according to *Beton-Bogen* (‘The Concrete Book’)/4/.

The reinforcement ratio φ is

Equation 37

$$\varphi = \frac{A_s}{A_b} = \frac{((b_b - \Delta b)/\Delta b) \cdot \pi \cdot \left(\frac{d}{2}\right)^2}{h_b \cdot b_b} = \frac{\pi \cdot d^2}{4 \cdot h_b \cdot \Delta b} \text{ for } b_b >> \Delta b$$

where

φ = reinforcement ratio (unitless)
A_s = cross-sectional area of reinforcement (mm^2)
A_b = cross-sectional area of concrete (mm^2).

The modulus of elasticity n is

Equation 38

$$n = \frac{E_s}{E_b}$$

where

n = modulus of elasticity (unitless)
E_s = coefficients of elasticity for steel (MPa)
E_b = coefficients of elasticity for concrete (MPa).

Compression stress of reinforcement

Equation 39

$$\sigma_s = \frac{\varepsilon_s \cdot E_s}{(1 + n \cdot \varphi) \cdot 100}$$

According to the Danish standard specification for concrete DS 411 /3/, crack width can be calculated by means of the formula:

Equation 40

$$w = 5 \cdot 10^{-5} \sigma_s \cdot \sqrt{a_w}$$

where

w = crack width (mm)
σ_s = reinforcement stress (MPa)
a_w = crack parameter (mm)

3.5.3 Calculating crack spacing

According to *Beton-Bogen* ('The Concrete Book') /4/, the smallest crack spacing can be calculated as:

Equation 41

$$l_m = \frac{a_w}{\pi}$$

where l_m = smallest crack spacing (mm).

Average crack spacing is calculated as

Equation 42

$$l_w = 1{,}5 \cdot l_m$$

where l_w = average crack spacing (mm).

This formula applies to $a_w < 2000$.

Equation 43

Total crack length is calculated as:

$$l_{tot} = (\frac{l_b \cdot 1000}{l_w} - 1) \cdot l_l + (\frac{l_l \cdot 1000}{l_w} - 1) \cdot l_b$$

where

l_{tot} = total crack length (m)
l_l = floor length (m)
l_b = floor width (m)

3.5.4 Calculating air transport through cracks

The following calculation of air transport through cracks is carried out according to Baker, Sharples & Ward /7/.

Volume flow through concrete deck is calculated by means of 'Cubic Law'

Equation 44

$$Q_b = \frac{l_{tot} \cdot w^3}{12 \cdot \mu} \cdot \frac{\Delta P \cdot 10^{-6}}{h_b}$$

where

Q_b = volume flow through concrete deck (m^3/s)
l_{tot} = total crack length (m)
w = crack width (mm)
ΔP = pressure differential above concrete deck (Pa)
h_b = concrete deck thickness (mm)
μ = dynamic viscosity of air in the soil system ($kg/m \cdot s$)

Volume flow q pr. m^2 floor area will be

Equation 45

$$q = \frac{Q_b}{A_g} = \frac{l_{tot} \cdot w^3}{8 \cdot \mu} \cdot \frac{\Delta P \cdot 10^{-6}}{h_b \cdot A_g}$$

where A_g = floor area (m^2) = $l_l \cdot l_b$

Mass balance around floor:

Diffusion through soil equals the sum of diffusion and convection through concrete:

Equation 46

$$N_1 \cdot D_L \frac{C_L - C_P}{x_1} = N_b \cdot D_L \frac{C_P - C_K}{x_b} + q \cdot C_P$$

where

N_1 = material constant for the soil stratum under the concrete deck, and x_1 is its thickness.
N_b = material constant of concrete deck and x_b is its thickness.
C_L = contaminant concentration in soil gas at the contamination mg/m^3).
C_P = soil gas concentration under concrete deck (mg/m^3).
C_K = the sum of convective and diffusive contributions to indoor air concentration (mg/m^3).
Q = volume flow (convection through floor cracks) pr. m^2 of floor area.
D_L = contaminant diffusion constant in air (m^2/s).

Mass balance of building:

The flux out of the buildings equals the sum of convection and diffusion through the concrete floor.

Equation 47

$$C_K \cdot L_h \cdot L_S = N_b \cdot D_L \frac{C_P - C_K}{x_b} + q \cdot C_P$$

where

L_h = ceiling height in building (m)
L_s = air renewal in building (pr. time)

Equations 46 and 47 give (two simultaneous equations):

Equation 48

$$C_K = \frac{\left(\frac{N_b\, D_L}{x_b} + q\right) C_L}{L_h\ L_S + \frac{N_b\, x_1\, L_h\, L_S}{x_b\, N_1} + \frac{N_b\ D_L}{x_b} + \frac{q\ x_1\, L_h\, L_S}{N_1\, D_L}}$$

If the soil gas concentration C_P is measured under floor, the contamination contribution $C_K^{'}$ to indoor air can be calculated using the following formula:

Equation 49

$$C'_K = \frac{C_P \cdot \left(\frac{N_b \cdot D_L}{x_b} + q \right)}{L_h \cdot L_S + \frac{N_b \cdot D_L}{x_b}}$$

where C'_K = contamination contribution calculated on the basis of the soil gas concentration measured (mg/m^3).

If there are several different soil strata under the concrete deck, a total material constant and equivalent thickness can be calculated for all strata:

Equation 50
$N_J = N_1$

Equation 51
$$x_J = x_1 + x_2 \cdot \frac{N_1}{N_2} + \ldots + x_n \frac{N_1}{N_n}$$

where

N_J = the equivalent material constant of all soil strata
x_J = the equivalent thickness of all soil strata

4 Data regarding types of soil, concrete parameters and building parameters

Tables 1 to 3 provide values for relevant parameters for various types of soil, concrete parameters and building parameters.

For soil parameters, four different types of soil are indicated: two types of mould representing the plough layer in fields (sandy loam and loam), and two types of soil representing soil below the plough layer/root zone. The water content provided corresponds to dry soil, defined as soil with natural drainage.

However, if the upper soil strata have growth (e.g. in connection with the two types of mould), water content might drop to a level corresponding to a water content which is inaccessible to plants.

For each parameter, Table 1 provides an interval in the top line and a typical value in the bottom line.

Table 1
Data regarding types of soil.

	Loam	Sandy loam	Clay	Sand
Porosity $V_l + V_v$, % (volume basis)	35-45 40	40-45 45	35-45 40	35-50 45
Water content V_v, % (volume basis)	25-35 30	15-35 35	20-40 30	5-35 15
Particle density, d, kg/l	2.6-2.7 2.65	2.5-2.6 2.6	2.7-2.8 2.7	2.6-2.7 2.65
Volumetric density, ρ, kg/l	1.4-1.8 1.7	1.4-1.7 1.6	1.5-1.8 1.8	1.4-1.7 1.7
Organic carbon content, f_{oc}, %, (weight basis)	1	2	0.1	0.1

In fissured clayey till there is a secondary porosity from the fissures, which gives a greater material constant depending on the water content of the matrix and fissures.

As regards concrete parameters for calculating the diffusive and convective contributions to indoor air, standard values are provided for a reinforced concrete deck as described in *Diffusionsforsøg, betongulve* (‘Diffusion tests, concrete floors’), issued by the National Housing and Building Agency /1/ and *Radonvejledningen* (‘The Radon Guidelines’) /2/.

Table 2
Data regarding concrete deck.

Material constant, N, unitless	0.002
Reinforcement bar spacing, Δb, mm	50
Reinforcement diameter, d_a, mm	3
Relative humidity, RF, %	60
Cement content, CM, kg/m^3	220
Water/cement ratio, V/C, unitless	0.67
Shrinkage time, t_s, days	7300
Coefficient of elasticity for steel, E_s, MPa	210,000
Coefficient of elasticity for concrete, E_b, MPa	20,000

As regards other building parameters, the following standard values may be used.

Table 3
Data regarding the building.

Pressure differential over concrete deck, Δ P, Pa	5
Air renewal, L_s, s^{-1}	$8.3 \cdot 10^{-5}$

Air renewal of $8.3 \cdot 10^{-5}$ s^{-1} corresponds to 0.3 time^{-1}.

S. 5.3.2 of the risk assessment for indoor airs mentions a dilution factor of 100. Using the following standard values:

Air renewal, L_s = $8.3 \cdot 10^{-5}$ s^{-1}
Air height, L_h = 2.3 m

Concrete thickness, h_b = 0.08 m
Pressure differential over concrete deck, ΔP = 5 Pa
a set of crack widths and lengths can be calculated for a floor of 100 m^2, ensuring a dilution factor of 100, cf. Equations 45 and 49.

Table 4
Crack width and crack length

Crack width (mm)	Crack length (m)
0.1	640
0.2	70
0.5	4.7
1.0	0.6

Table 5
List of symbols.

Symbol	Explanation	Unit
α	Auxiliary parameter	unitless
α_w	Crack parameter	mm
β	Auxiliary parameter	unitless
ε_c	Base shrinkage	%
ε_s	Shrinkage strain	%
μ	Dynamic viscosity of air	kg/m/s
ρ	Soil density	kg/l
σ_s	Reinforcement tension	MPa
Φ_n	Neutral acid fraction	unitless
φ	Reinforcement ratio	unitless
A	Area	m^2
A_{cef}	Active concrete area subject to tensile force	mm^2
a_w	Crack parameter	mm
b	Contaminated area width	m
b_b	Considered width of concrete deck	mm
Δb	Reinforcement-bar spacing	mm
C_i	Diffusive contamination contribution to indoor air	mg/m^3
C_L	Soil gas concentration at source	mg/m^3
$C_{L,max}$	Saturated vapour concentration	mg/m^3
C_K	Convective and diffusive contamination contributions	mg/m^3
C'_K	Contamination contribution calculated on the basis of registered soil gas concentration	mg/m^3
CM	Cement content	kg/m^3
C_o	Background concentration	mg/m^3
C_p	Soil gas concentration under concrete deck	mg/m^3
C'_P	Soil gas concentration measured	mg/m^3
C_T	Total soil concentration	mg/kg
C_u	Diffusive contamination contribution to outdoor air	mg/m^3
d	Particle density	kg/l
d_a	Reinforcement diameter	mm
d_w	Crack-determining reinforcement diameter	mm
D_L	Diffusion coefficient	m^2/s
E_b	Coefficient of elasticity for concrete	MPa
E_s	Coefficient of elasticity for steel	MPa
f_L	Relative proportion of soil gas contaminant in relation to total content	unitless
f_{oc}	Organic carbon content	relative

Symbol	Explanation	Unit
		weight proportion
h	Mixing height in atmosphere.	m
h_b	Thickness of concrete deck	mm
J	Flux (evaporation)	$mg/m^2/s$
J_{max}	Amount adsorbing to the organic fraction of soil particles at equilibrium with saturated solution in water	mg/kg
k	Constant dependent on reinforcement	unitless
k_b	Coefficient taking into account influences from composition of concrete	unitless
K_D	Soil/water proportion	l/kg
k_d	Coefficient taking into account influences from dimensioning	unitless
k_f	Coefficient taking into account influences from shrinkage time	unitless
K_{oc}	Organic carbon/water proportion	l/kg
K_{ow}	Octanol/water proportion	unitless
l	Contaminated area length	m
l_b	Floor width	m
L_h	Ceiling height of building	m
l_l	Floor length	m
l_m	Smallest crack spacing in concrete	mm
L_s	Air renewal in building	s^{-1}
l_{tot}	Total crack length in concrete	m
l_w	Average crack spacing in concrete	mm
m	Molecular weight	g/mol
M_J	Actual volume adsorbed to soil particles	mg/m^3 soil
$M_{J.max}$	Maximum volume adsorbed to soil particles	mg/m^3 soil
M_L	Actual volume in soil gas	mg/m^3 soil
$M_{L.max}$	Maximum volume in soil gas	mg/m^3 soil
M_V	Actual volume in soil water	mg/m^3 soil
$M_{V.max}$	Maximum volume in soil water	mg/m^3 soil
n	Elasticity modulus	unitless
N_n	Material constant for strata no. n	unitless
N_J	Equivalent material constant	unitless
Δ P	Pressure differential above concrete deck	Pa
p	Partial pressure	N/m^2
pK_a	Acid dissociation constant	unitless
Q	Mass flow velocities	mg/s
Q_b	Volume flow through concrete deck	m^3/s
q	Volume flow through concrete deck pr. m^2	m/s
r	Equivalent radius of construction	mm
R	Gas constant	$J/mol \cdot K$
RF	Relative humidity	%
S	Solubility	mg/m^3
T	Temperature	K
t_0	Auxiliary parameter	unitless
t_s	Shrinkage time	days
v	Wind velocity	m/s
v/c	Water/cement ratio	unitless
V_J	Relative volumetric proportion of soil particles in soil	%
V_L	Relative volumetric proportion of air in soil	%
V_V	Relative volumetric proportion of water in soil	%
w	Concrete crack width	mm
X_n	Thickness of diffusion- inhibiting strata no. n	m
x_J	Equivalent thickness	m

References

/1/ The National Housing and Building Agency, 1992: *Diffusionsforsøg, betongulve* (‘Diffusion tests, concrete floors’).

/2/ The National Housing and Building Agency, 1993: *Radon og Nybyggeri* (‘Radon and New Buildings’)

/3/ *Dansk Ingeniørforenings norm for betonkonstruktioner* (‘Danish standard specifications for concrete constructions’) , 1984. Dansk Standard DS411.

/4/ Herholdt, A.D., Justesen, C.F.P., Nepper Christensen, P. & Nielsen, A. 1985: *Beton-Bogen* (‘The Concrete Book’).

/5/ Mackay, D. and Paterson, S.: *Calculating Fugacity*, Environmental Science and Technology, Vol. 15, No 9, 1981.

/6/ The Environmental Protection Agency. Project No. 20, 1996: *Kemiske stoffers opførsel i jord og grundvand* (‘Chemical substance behaviour in soil and groundwater’)

/7/ Baker, P.H., Sharples, S. & Ward I.C.: *Air Flow Through Cracks". Building and environment, vol.22, no.4 1987.*
Cracks. Building and Environment, vol. 22, no. 4, 1987.

/8/ Millington, R.J.: *Gas Diffusion in Porous Media*, Science, 130, 1959.

/9/ Lugg, G.A.: *Diffusion Coefficients of Some Organic and Other Vapours in Air*, Analytical Chemistry, 40, 1968.

Example of calculation of evaporation of volatile substances from soil

1 Background

The following example, cf. Figure 1, features a calculation of the contributions of trichloroethylene contamination to indoor air and outdoor air concentrations. The equations are numbered in accordance with Appendix 5.3.

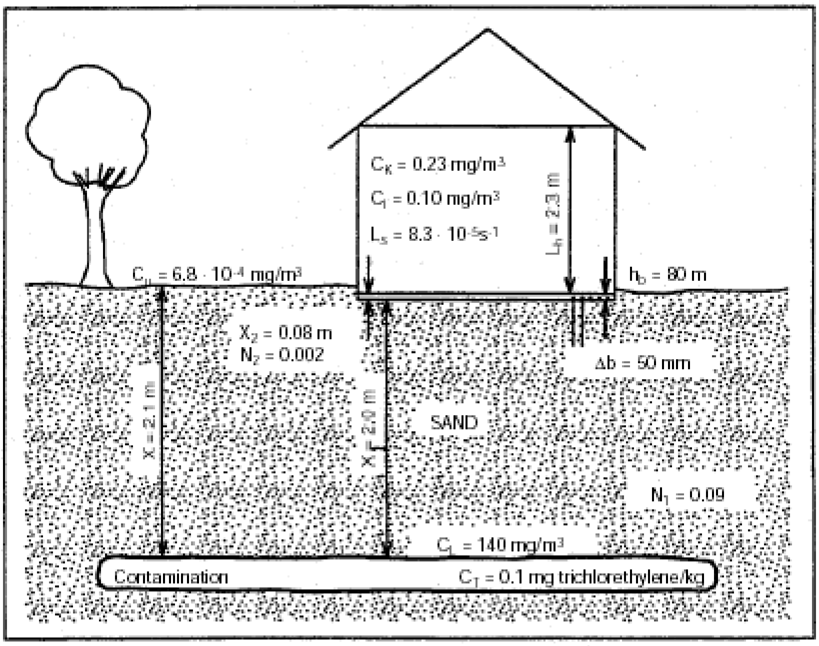

Figure 1
Calculation example, contributions to outdoor air and indoor air.

The following two tables show the standard values, imaginary investigation values and chemical constants used in connection with calculation.

Table 1
Standard values and investigation values

	Standard value	Investigation value
Relative volumetric proportion of air, V_L	0.30	
Relative volumetric proportion of water, V_V	0.15	
Relative volumetric proportion of soil, V_J	0.55	
Temperature, T	298 K = 25° C	
Soil particle density, d	2.65 kg/l	
Soil trichloroethylene concentration, C_T		0.1 mg/kg
Soil density, ρ	1.7 kg/l	
Soil content of organic substances, f_{oc}	0.002	
Sand strata thickness, outdoors, X		2.1 m
Sand strata thickness, under floor, X_1		2.0 m
Concrete deck thickness, X_2		0.08 m
Material constant for concrete, N_2	0.002	
Building ceiling height, L_h		2.3 m
Building air renewal, L_s	$8.3 \cdot 10^{-5}$ s^{-1}	
Pressure differential over concrete deck, ΔP	5 Pa	
Concrete deck thickness, h_b		80 mm
Reinforcement bar spacing, Δb	50 mm	
Reinforcement diameter, d_a	3 mm	
Relative humidity, RF	60 %	
Cement content CM	220 kg/m^3	
Water/cement ratio, V/c	0.67	
Shrinkage time, t_s	7,300 days	
Dynamic viscosity of air, μ	$1.8 \cdot 10^{-5}$ (kg/m) · s	
Floor length		10 m
Floor width		10 m

The chemical constants can be found in tabular form in Appendix 5.5

Table 2
Chemical constants.

Trichloroethylene partial pressure, p	9,900 N/m^2
Trichloroethylene molecular weight, m	131.39 g/mol
Gas constant, R	8.314 J/mol·K
Trichloroethylene solubility, S	1,400,000 mg/m^3
Trichloroethylene diffusion coefficient, D_L	$8.8 \cdot 10^{-6}$ m^2/s
Trichloroethylene octanol-water ratio, K_{ow}	$10^{2,53}$ l/kg

2 Calculations

2.1 Phase distribution in soil

The total volume of soil can be seen as the sum of the volumes of the soil phases, see Equation 1.

Equation 1

$$V_L + V_V + V_J = 1$$

where: V_L =relative volumetric proportion of air in soil
V_V =relative volumetric proportion of water in soil
V_J =relative volumetric proportion of soil particles in soil

Maximum trichloroethylene in one cubic metre (1 m^3) of soil distributed on the three phases of soil can be calculated as follows, see Equations 2-10:

In the air phase of soil (soil gas):

Equation 2

$$M_{L,\,max} = V_L \cdot C_{L,\,max} = 0.30 \cdot 525{,}000 \text{ mg/m}^3 = 158{,}000 \text{ mg/m}^3$$

where: $M_{L,max}$ = maximum amount of trichloroethylene in soil gas (mg/m^3 soil volume)
$C_{L,max}$ = saturated vapour concentration of contaminant (mg/m^3 soil gas).

$C_{L,max}$ can be calculated on the basis on the partial pressure of trichloroethylene partial pressure by means of the law of ideal gases:

Equation 3

$$C_{L,max} = \frac{p \cdot m \cdot 10^3}{R \cdot T} = \frac{9900 \text{ N/m}^2 \cdot 131.39 \text{ g/mol} \cdot 10^3}{8.314 \text{ J/mol/K} \cdot 298 \text{K}} = 557{,}000 \text{ mg/m}^3$$

where: p =trichloroethylene partial pressure (9,900 N/m^2)
m =trichloroethylene molecular weight (131.39 g/mol)
R =gas constant (8.314 J/mol · K)
T =temperature (298 K = 25° C)

In the water phase of soil (soil water):

Equation 4

$$M_{V,max} = V_V \cdot S = 0.15 \cdot 1{,}400{,}000 \text{ mg/m}^3 = 210{,}000 \text{ mg/m}^3$$

where: $M_{V,max}$ = maximum amount of trichloroethylene in soil water (mg/m^3 soil volume)
S = trichloroethylene solubility in water (1,400,000 mg/m^3 soil water).

In the partial phase of soil:

Equation 9

$M_{J,max} = V_J \cdot d \cdot K_{oc} \cdot f_{oc} \cdot S$ (mg/m^3)

$= 0.55 \cdot 2.65$ kg/l $\cdot 10^{1,79} \cdot 0.002 \cdot 1{,}400{,}000$ mg/m^3 = 252,000 mg/m^3

where: $M_{J,max}$ = maximum amount of trichloroethylene, which has adsorbed to the organic fraction of the soil particles (mg/m^3 soil volume)

d = soil particle density (2.65 kg/l)

K_{oc} = trichloroethylene ratio between organic carbon and water (1/kg)

f_{oc} = soil content of organic carbon (0.002)

The trichloroethylene ratio between carbonate and water can be estimated on the basis of the octanol/water ratio K_{ow}, (equation 10a).

$\log K_{oc} = 1.04 \cdot \log K_{ow} - 0.84 = 1.04 \cdot 2.53 - 0.84 = 1.79$

Equation 10a

The soil's maximum capacity for trichloroethylene (immediately preceding NAPL) will then be:

$M_{L,max} + M_{V,max} + M_{J,max}$

= 158,000 mg/m^3 + 210,000 mg/m^3 + 252,000 mg/m^3

= 620,000 mg/m^3

Based on the above-mentioned assumption that the relative distribution among the three soil phases is independent of total soil concentration, the distribution of trichloroethylene in the three phases of soil can be calculated.

The following applies to the air phase of soil

Equation 11

$$f_L = \frac{M_{L,max}}{M_{L,max} + M_{V,max} + M_{J,max}} = \frac{M_L}{M_L + M_V + M_J} = \frac{167{,}000}{629{,}000} = 0.266$$

where:

f_L = relative amount of trichloroethylene in soil gas in relation to total soil content (calculated pr. m^3 soil).

M_L, M_V, M_J = actual amount of trichloroethylene in each of the three phases (mg/m^3 soil).

With a total soil concentration C_T (0.1 mg trichloroethylene/kg soil volume) the amount of trichloroethylene in air M_L can be established:

Equation 13

$M_L = f_L \cdot C_T \cdot \rho \cdot 10^3 = 0.25 \cdot 0.1$ mg/kg $\cdot$ 1.7 kg/l $\cdot 10^3$ = 43.3 mg/m^3 soil volume

where: C_t = trichloroethylene concentration in soil (0.1 mg/kg)

ρ = soil density (1.7 kg/l)

The trichloroethylene concentration in soil gas, C_L, is now calculated on the basis of trichloroethylene concentration in soil, C_T, see equation 14.

Equation 14

$$C_L = \frac{M_L}{V_L} = \frac{45.2 \ \text{mg/m}^3 \ \text{jordvolumen}}{0.3} = 150 \, \text{mg/m}^3 \ \text{poreluft}$$

C_L does not exceed $C_{L,max}$, which means that there is no NAPL. If free C_L exceeds $C_{L,max}$, there is NAPL and $C_{L,max}$ is used in subsequent calculations.

As the calculated soil gas concentration under building is more than 100 times greater the evaporation criteria (0.001 mg/m^3) and is more than 10 times greater than the evaporation criteria in the open-air area, soil gas measuring can be carried out. If a sufficient number of soil gas measurements show that the soil gas concentration is less than 0.1 µg/m^3 under the building or less than 0.01 µg/m^3 in the open-air area, the site can be cleared.

If the soil gas concentrations measured are more than 10 and 100 times greater, respectively, than the evaporation criterion, calculations are made of the diffusive contribution to outdoor air, cf. Section 2.2 and the diffusive and convective contributions to indoor air, cf. Sections 2.3 and 2.4.

2.2 Diffusive contribution to the contamination concentrations in outdoor air

Equation 15

$$J = -N \cdot D_L \frac{C_0 - C_L}{X} = -0.09 \cdot 8.8 \cdot 10^{-6} \text{m}^2/\text{s} \cdot \frac{0 \, \text{mg/m}^3 - 140 \, \text{mg/m}^3}{2.1 \text{m}}$$

$$J = 5.4 \cdot 10^{-5} \text{ mg/(m} \cdot \text{ s)}$$

where: J = flux (evaporation) (mg/(m^2 · s))
N = material constant (unitless)
D_L = diffusion coefficient of trichloroethylene in air (8.8 · 10^{-6} m^2/s)
X = depth corresponding to concentration C_L (2.1 m)
C_o = background concentration at the site (mg/m^3) is set at 0, as it is much smaller than C_L.

The material constant N for sand is calculated as:

Equation 18

$$N = V_L^{3,33} / (V_L + V_v)^2 = 0.30^{3,33}/(0.30+0.15)^2 = 0.09$$

Equation 21

$$C_u = \frac{J}{v \cdot \ 0{,}08} = \frac{5{,}7 \cdot 10^{-5} \ \text{mg/(m}^2 \cdot \text{s)}}{2 \ \text{m/s} \cdot \ 0{,}08} = 3{,}5 \cdot 10^{-4} \ \text{mg/m}^3$$

where: C_u = diffusive trichloroethylene contribution to outdoor
v = wind velocity (1 m/s).

The diffusive contribution to outdoor air is then 0.00068 mg/m^3, which is less than the evaporation criterion of 0.001 mg/m^3.

2.3 Diffusive contribution to indoor air contamination concentrations

Equation 17

$$J = \frac{-(N_1 \cdot N_2) \cdot D_L \cdot (C_o - C_L)}{N_2 \cdot X_1 + N_1 \cdot X_2}$$

$$J = \frac{-(0{,}09 \cdot 0{,}002) \cdot 8{,}8 \cdot 10^{-6}\ m^2/s \cdot (0\ mg/m^3 - 150\ mg/m^3)}{0{,}002 \cdot 2{,}0\ m + 0{,}09 \cdot 0{,}08\ m}$$

$$J = 1{,}47 \cdot 10^{-5}\ mg/(m^2 \cdot s)$$

where: J = flux (evaporation) (mg/(m^2 · s))
N_1 = material constant for sand (0.09)
N_2 = material constant for concrete (0.002)
D_L = trichloroethylene diffusion coefficient in air ($8.8 \cdot 10^{-6}$ m^2/s)
X_1 = thickness of sand strata (2.0 m)
X_2 = thickness of concrete (0.08 m)
C_o = the background concentration at the site (mg/m^3) is set at 0, as it is a lot smaller than C_L.

Equation 18
$N_1 = V_L^{3,33} / (V_L + V_v)^2 = 0.3^{3,33}/(0.3+0.15)^2 = 0.09$,
$N_2 = 0.002$ corresponding to environmentally neutral concrete.

Equation 24

$$C_i = \frac{J}{L_h \cdot L_s} = \frac{1.47 \cdot 10^{-5}}{2.3\ m \cdot 1.4 \cdot 10^{-4}\ s^{-1}} = 0.061\ mg/m^3$$

where: C_i = diffusive trichloroethylene contribution to indoor air (mg/m^3)
L_h = building ceiling height (2.3 m)
L_s = air renewal in building ($8.3 \cdot 10^{-5}$ s^{-1})

This establishes the diffusive contribution to the indoor air as 0.10 mg/m^3.

2.4 Convective contribution through concrete deck to indoor air contamination concentration

2.4.1 Calculating crack lengths and widths

The floor made of 8 centimetres of reinforced environmentally neutral concrete with 20 reinforcement bars, each with 3 mm tentor steel pr. 1000 mm; this corresponds to a concrete deck in accordance with *Radonvejledningen* ('The Radon Guidelines') /3/.

Equation 27
$d_w = k \cdot d_a = 1 \cdot 3\ mm = 3\ mm$

where d_w = crack-determining diameters of reinforcement bars
d_a = nominal reinforcement diameter (3 mm)
k = 1, due to the fact that the reinforcement is tentor steel

Equation 29

$$a_w = \frac{h_b \cdot \Delta b}{d_w} = \frac{80\ \text{mm} \cdot 50\ \text{mm}}{3\ \text{mm}} = 1333\ \text{mm}$$

where a_w = crack parameter (mm)
h_b = thickness of concrete deck (80 mm)
Δb = reinforcement bar spacing (50 mm)

The free shrinkage strain ε_s can be calculated as

Equation 30

$$\varepsilon_s = \varepsilon_c \cdot k_b \cdot k_d \cdot k_t = 0.0333\% \cdot 1.035 \cdot 0.866 \cdot 0.989 = 0.0295\ \%$$

where ε_s = shrinkage strain (%)
ε_c = base shrinkage (%), see Equation 30
k_b = coefficient taking into account influences from the composition of concrete (unitless), see Equation 31
k_d = coefficient taking into account influences from geometry (unitless), see Equation 33
k_t = coefficient taking into account influences from shrinkage time (unitless), see Equation 34

Base shrinkage can be calculated as

Equation 31

$$\varepsilon_c = \frac{0.089 \cdot (100 - \text{RF})}{167 - \text{RF}} = \frac{0.089 \cdot (100 - 60)}{167 - 60} = 0.0333\ \%$$

where RF = relative humidity (60 %)

k_b is calculated on the basis of the composition of concrete

Equation 32

$$k_b = 0.007 \cdot \text{CM} \cdot (\text{v/c} + 0.333) \cdot \text{v/c}$$
$$k_b = 0.007 \cdot 220 \cdot (0.67 + 0.333) \cdot 0.67 = 1.035$$

where CM = cement content (220 kg/m^3)
v/c = water/cement ratio (0.67)

The equivalent radius r and k_d are calculated by means of the following formulae:

Equation 33

$$r = 2 \cdot h_b = 2 \cdot 80\ \text{mm} = 160\ \text{mm}$$

where r = equivalent construction radius (mm)

Equation 34

$$k_d = \frac{0.25 \cdot (852 + r)}{132 + r} =$$

Influence of time:

Equation 35

$$k_t = \frac{t_s^{\alpha}}{t_s^{\alpha} + t_0} = \frac{7300^{0,96}}{7300^{0,96} + 57.4} = 0.989$$

where t_s = shrinkage time (7,300 days)
t_o, α and β are auxiliary parameters (unitless)

Equation 36

$$t_0 = 9 \cdot (\sqrt{10})^{\alpha \bullet \beta} = 9 \cdot (\sqrt{10})^{0,96 \bullet 1,68} = 57.4$$

$$\alpha = 0.75 + 0.125 \cdot \beta = 0.75 + 0.125 \cdot 1.68 = 0.96$$

$$\beta = \frac{\ln(0.02 \cdot r)}{\ln 2} = \frac{\ln(0.02 \cdot 160)}{\ln 2} = 1.68$$

2.4.2 Calculating reinforcement tension

The calculation is carried out in accordance with *Beton-Bogen* (‘The Concrete Book’) /1/.

The reinforcement ratio φ is:

Equation 37

$$\varphi = \frac{A_s}{A_b} = \frac{\pi\pi \cdot d_a^2}{4 \cdot h_b \cdot \Delta b} = \frac{\pi\pi \cdot (3\,\mathrm{mm})^2}{4 \cdot 80\,\mathrm{mm} \cdot 50\,\mathrm{mm}} = 0.00177$$

where φ = reinforcement ratio (unitless)
A_s = cross-section area of reinforcement (28.27 mm^2)
A_b = cross-section area of concrete (16,000 mm^2).

Elastic strain n is

Equation 38

$$n = \frac{E_s}{E_b} = \frac{210{,}000\,\mathrm{MPa}}{20{,}000\,\mathrm{MPa}} = 10.5$$

where n = elastic strain (unitless)
E_s = coefficients of elasticity for steel (210,000 MPa)
E_b = coefficients of elasticity for concrete (20,000 MPa).

Compression stress of reinforcement

Equation 39

$$\sigma_s = \frac{\varepsilon_s \cdot E_s}{(1 + n \cdot \varphi) \cdot 100} = \frac{0.0295 \cdot 210{,}000}{(1 + 10.5 \cdot 0.00177) \cdot 100} = 60.8\,\mathrm{MPa}$$

According to the standard specifications of concrete DS 411 /2/ the crack width can be calculated by means of the formula:

Equation 40

$$w = 5 \cdot 10^{-5} \sigma_s \cdot \sqrt{a_w} = 5 \cdot 10^{-5} \cdot 60.8 \cdot \sqrt{1333} = 0.111\text{mm}$$

where w = crack width (mm)
σ_s = reinforcement tension (60.8 MPa)
a_w = crack parameter (1,667 mm)

The formula then applies to $a_w < 2{,}000$.

2.4.3 Calculating crack spacing

According to *Beton-Bogen* (‘The Concrete Book’) /1/ the smallest crack spacing can be calculated as

Equation 41

$$l_m = \frac{a_w}{\pi} = \frac{1333 \text{ mm}}{\pi} = 424 \text{ mm}$$

where l_m = smallest crack spacing (mm).

The average crack spacing is calculated as:

Equation 42

$$l_w = 1.5 \cdot l_m = 1.5 \cdot 424 \text{ mm} = 636 \text{ mm}$$

where l_w = average crack spacing (mm).

Total crack length is calculated as:

Equation 43

$$l_{tot} = \left(\frac{l_b \cdot 1000}{l_w} - 1\right) \cdot l_l + \left(\frac{l_l \cdot 1000}{l_w} - 1\right) \cdot l_b$$

$$l_{tot} = \left(\frac{10 \text{ m} \cdot 1000}{636 \text{ mm}} - 1\right) \cdot 10 \text{ m} + \left(\frac{10 \text{ m} \cdot 1000}{636 \text{ mm}} - 1\right) \cdot 10 \text{ m}$$

$$l_{tot} = 294 \text{ m}$$

where l_{tot} = total crack length (m)
l_l = floor length (10 m)
l_b = floor width (10 m)

Volume flow q pr. m^2 floor area will be:

2.4.2 Calculating air transport through cracks

Equation 45

$$q = \frac{l_{tot} \cdot w^3}{12 \cdot \mu} \cdot \frac{\Delta P \cdot 10^{-6}}{h_b \cdot A_g} = \frac{294 \text{ m} \cdot (0.111)^3}{12 \cdot 1.76 \cdot 10^{-5} \text{kg / m s}} \cdot \frac{10 \text{ Pa} \cdot 10^{-6}}{80 \text{ mm} \cdot 100 \text{ m}^2}$$

$$q = 2.38 \cdot 10^{-6} \text{ m}^3 / \text{s} / \text{m}^2$$

where q = volume flow pr. m^2 floor area ($(m^3/s)/m^2$)

ΔP = pressure differential over concrete deck (5 Pa)
μ = dynamic viscosity of gas ($1.8 \cdot 10^{-5}$ (kg/m) · s)
A_g = floor area (100 m^2), i.e. $l_l \cdot l_b$
w = crack width (0.111 mm)
h_b = concrete-deck thickness (80 mm)
l_{tot} = total crack length (294 m)

The concentration over floor C_k can be calculated as follows:

Equation 48

$$C_K = \frac{\left(\frac{N_b\, D_L}{x_b} + q\right) C_L}{L_h\, L_S + \frac{N_b\, x_1\, L_h\, L_S}{x_b\, N_1} + \frac{N_b\, D_L}{x_b} + \frac{q\, x_1\, L_h\, L_S}{N_1\, D_L}}$$

N_1 =material constant for sand = 0.09
D_L =trichloroethylene diffusion coefficient in air = $8.8 \cdot 10^{-6}$ m^2/s
C_L = concentration of contaminants in soil gas at a contamination = 140 mg/m^3 soil gas
x_1 =thickness of sand stratum under floor = 2.0 m
N_b =material constant for concrete = 0.002
x_b =thickness of concrete deck = 0.08 m
q =$1.16 \cdot 10^{-6}$ (m^3/s)/m^2
L_S =air renewal in building = $8.3 \cdot 10^{-5}$ s^{-1}
L_h =building ceiling height 2.3 m

When these values are substituted into the equation the result is:

Equation 48

$$C_K = \frac{\left(\frac{0.002 \cdot 8.8 \cdot 10^{-6}}{0.08} + 1.16 \cdot 10^{-6}\right) \cdot 140 \text{mg} \cdot \text{m}^{-3}}{2.3 \cdot 8.3 \cdot 10^{-5} + \frac{0.002 \cdot 2.0 \cdot 2.3 \cdot 8.3 \cdot 10^{-5}}{0.08 \cdot 0.09} + \frac{0.02 \cdot 8.8 \cdot 10^{-6}}{0.08} + \frac{1.16 \cdot 10^{-6} \cdot 2.0 \cdot 2.3 \cdot 8.3 \cdot 10^{-5}}{0.09 \cdot 8.8 \cdot 10^{-6}}}$$

The calculated diffusive contribution (C_i = 0.10 mg/m^3) as well as the total contribution (C_k = 0.23 mg/m^3) are greater than the evaporation criterion for trichloroethylene of 0.001 mg/m^3.

As the evaporation criterion for indoor air has been exceeded, the indoor air must be investigated; alternatively remedial measures to ensure an acceptable indoor air must be carried out.

If soil gas measuring has been carried out, the relevant values are to be used in calculating equation 15 or 17 in Section 2.2 and 2.3, being substituted as C_L while taking into account the distance to the measuring point X_n.

References

/1/ Herholdt, A.D., Justesen, C.F.P., Nepper Christensen, P. & Nielsen, A. 1985: *Beton-Bogen* ('The Concrete Book').

/2/ *Dansk Ingeniørforenings norm for betonkonstruktioner* ('Danish standard specifications for concrete constructions'), 1984. Dansk Standard DS411.

/3/ The National Housing and Building Agency, 1993: *Radon og Nybyggeri* ('Radon and New building')

Physical and chemical data

In connection with calculations of indoor air or outdoor air, soil gas concentration can be estimated on the basis of analyses of soil or water samples by means of the principle of fugacity.

For the substance groups:

- monoaramtic hydrocarbons,
- polycyclic aromatic hydrocarbons,
- aliphatic hydrocarbons,
- chlorinated aliphatics

the chemical data for:

- molecular weight
- vapour pressure
- water solubility
- octanol-water distribution coefficient

are listed in the following.

For phenols, the chemical substances mentioned above as well as the acid dissociation constant are listed.

The chemical constants

- molecular weight
- vapour pressure
- water solubility
- octanol-water distribution coefficient
- acid-dissociation constant

are from *Miljøstyrelsens Projekt om jord og grundvand* ('The Environmental Protection Agency Project on soil and groundwater') No. 20. /1/.

Diffusion coefficients are taken from Lugg /2/ or estimated on the basis of the following formula from *Miljøstyrelsens Projekt om jord og grundvand* ('The Environmental Protection Agency Project on soil and groundwater'), formula 1, No. 20. /1/.

$$D_2 = D_1 \sqrt{\frac{m_1}{m_2}}$$

where

D_1 and D_2 = contaminant diffusion coefficients in air (m^2/s).
m_1 and m_2 = contaminant molecular weights (g/mol).

Table 1
Chemical data for monoaromatic hydrocarbons

Substance name	Molecular weight	Vapour pressure	Water solubility	Octanol-water coefficient	Diffusion coefficient in air
	m	p	S	log K_{ow}	D_L
	g/mol	Pa	mg/l	-	m^2 /s
Benzene	78.1	12700	1760	2.1	$9.3 \cdot 10^{-6}$
Toluene	92.1	3800	550	2.7	$8.5 \cdot 10^{-6}$
o-Xylene	106.2	880	180	3.1	$7.3 \cdot 10^{-6}$
m-Xylene	106.2	1110	160	3.2	$6.9 \cdot 10^{-6}$
p-Xylene	106.2	1170	200	3.2	$6.7 \cdot 10^{-6}$
1,2,3-Trimethylbenzene	120.2	202	66	3.6	$7.1 \cdot 10^{-6}$ *)
1,3,5-Trimethylbenzene	120.2	328	50-173	3.4	$7.1 \cdot 10^{-6}$ *)
1,2,4-Trimethylbenzene	120.2	271	66	3.6	$7.1 \cdot 10^{-6}$*)
Ethylbenzene	106.2	1270	170	3.2	$7.6 \cdot 10^{-6}$
1-Ethyl-2-methylbenzene	120.2	330	40-93	3.5	$7.1 \cdot 10^{-6}$ *)
1-Ethyl-4-methylbenzene	120.2	493	95	3.6	$7.1 \cdot 10^{-6}$ *)

*) Estimated on the basis of the diffusion coefficient of the ethylbenzene and formula 1.

Table 2
Chemical data for polycyclic aromatic hydrocarbons

Substance name	Mole-cular weight	Vapour pressure	Water solubility	Octanol-water distribution coefficient	Diffusion coefficient in air
	m	p	S	log K_{ow}	D_L
	g/mol	Pa	mg/l	-	m^2/s
Naphthalene	128.2	10.4	31.0	3.36	$6.9 \cdot 10^{-6}$ *)
1-methylnaphthalene	142.2	8.8	28.5	3.87	$6.5 \cdot 10^{-6}$ *)
2-methylnaphthalene	142.2	9.0	25.4	3.86	$6.5 \cdot 10^{-6}$ *)
Biphenyl	154.2	1.3	7.5	4.1	$6.3 \cdot 10^{-6}$ *)
Acenaphthylene	154.2	0.90	3.93	4.1	$6.3 \cdot 10^{-6}$ *)
Acenaphthene	154.2	0.30	3.42	3.92	$6.3 \cdot 10^{-6}$ *)
Fluorene	166.2	0.090	1.98	4.18	$6.0 \cdot 10^{-6}$ *)
Phenanthrene	178.2	0.016	1.2	4.57	$5.8 \cdot 10^{-6}$ *)
Anthracene	178.2	$1.4 \cdot 10^{-3}$	0.041	4.54	$5.8 \cdot 10^{-6}$ *)
Fluoranthene	202.3	$1.3 \cdot 10^{-3}$	0.21	5.22	$5.5 \cdot 10^{-6}$ *)
Pyrene	202.3	$6.1 \cdot 10^{-4}$	0.14	5.18	$5.5 \cdot 10^{-6}$ *)
Benzyl(a)anthracene	228.3	$2.7 \cdot 10^{-5}$	0.014	5.61	$5.2 \cdot 10^{-6}$ *)
Chrysene	228.3	$8.4 \cdot 10^{-7}$	$2.0 \cdot 10^{-3}$	5.91	$5.2 \cdot 10^{-6}$ *)
Benzyl(b) fluoranthene	252.3	$5.0 \cdot 10^{-7}$	$1.5 \cdot 10^{-3}$	6.57	$4.9 \cdot 10^{-6}$ *)
Benzyl(k) fluoranthene	252.3	$1.3 \cdot 10^{-8}$	$8.0 \cdot 10^{-4}$	6.84	$4.9 \cdot 10^{-6}$ *)
Benzyl(e)pyrene	252.3	$7.4 \cdot 10^{-7}$	$4.0 \cdot 10^{-3}$	6.44	$4.9 \cdot 10^{-6}$ *)
Benzyl(a)pyrene	252.3	$7.3 \cdot 10^{-7}$	$3.8 \cdot 10^{-5}$	6.50	$4.9 \cdot 10^{-6}$ *)
Benzyl(g,h,i)perylene	276.3	$1.3 \cdot 10^{-8}$	$2.6 \cdot 10^{-4}$	6.90	$4.7 \cdot 10^{-6}$ *)
Dibenzyl(a,h) anthracene	278.4	$3.7 \cdot 10^{-10}$	$5.0 \cdot 10^{-4}$	6.50	$4.7 \cdot 10^{-6}$ *)

*) Estimated on the basis of the diffusion coefficient of the ethylene benzene and formula 1.

Table 3
Chemical data for aliphatic hydrocarbons.

Substance name	Mole-cular weight	Vapour pressure	Water solubility	Octanol-water distribution coefficient	Diffusion coefficient in air
	m	p	S	log K_{ow}	D_L
	g/mol	Pa	mg/l	-	m^2/s
Methane	16.0	$2.8 \cdot 10^7$	24.2	1.09	$1.8 \cdot 10^{-5}$*1)
Ethane	30.1	$4.0 \cdot 10^6$	61.5	1.81	$1.3 \cdot 10^{-5}$ *1)
Propane	44.1	$9.5 \cdot 10^5$	66.8	2.36	$1.1 \cdot 10^{-5}$ *1)
n-Butane	58.1	$2.5 \cdot 10^5$	60.8	2.89	$9.4 \cdot 10^{-6}$ *1)
n-Pentane	72.2	$7.0 \cdot 10^4$	40.6	3.62	$8.4 \cdot 10^{-6}$
n-Hexane	86.2	$2.1 \cdot 10^4$	12.8	4.11	$7.3 \cdot 10^{-6}$
n-Heptane	100.2	$6.2 \cdot 10^3$	3.10	4.66	$6.2 \cdot 10^{-6}$
n-Octane	114.2	$1.8 \cdot 10^3$	$7.2 \cdot 10^{-1}$	5.18	$5.8 \cdot 10^{-6}$ *3)
Cyclopentane	70.1	$4.2 \cdot 10^4$	156	3.00	$8.6 \cdot 10^{-6}$ *1)
Cyclohexane	84.2	$1.3 \cdot 10^4$	55	3.44	$7.4 \cdot 10^{-6}$ *2)
Cycloheptane	98.2	$2.9 \cdot 10^3$	30	3.91	$6.9 \cdot 10^{-6}$ *2)
Cyclo-octane	112.2	$7.5 \cdot 10^2$	7.9	4.47	$5.8 \cdot 10^{-6}$ *3)
1-Hexene	84.2	$2.5 \cdot 10^4$	50	3.39	$7.4 \cdot 10^{-6}$ *2)
1-Octene	112.2	$2.4 \cdot 10^4$	3.4	4.57	$5.8 \cdot 10^{-6}$ *3)

*1) Estimated on the basis of the diffusion coefficient of pentane and formula 1
*2) Estimated on the basis of the diffusion coefficient of hexane and formula 1
*3) Estimated on the basis of the diffusion coefficient of heptane and formula 1

Table 4
Chemical data for chlorinated aliphatics.

Substance name	Molecular weight	Vapour pressure	Water solubility	Octanol-water distribution coefficient	Diffusion coefficient in air
	m	p	S	log K_{ow}	D_L
	g/mol	Pa	mg/l	-	m^2/s
Chloromethane	50.49	570,000	5,235	0.91	$1.4 \cdot 10^{-6}$ *1)
Dichloromethane	84.94	48,300	13,200	1.25	$10.4 \cdot 10^{-6}$
Trichloromethane	119.38	26,244	8,700	1.97	$8.8 \cdot 10^{-6}$ *1)
Tetrachloromethane	153.82	15,250	780	2.64	$8.3 \cdot 10^{-6}$
Chloroethane	64.52	133,000	5,700	1.43	$1.1 \cdot 10^{-5}$ *2)
1,1-Dichloroethane	98.96	30,260	4,767	1.79	$9.2 \cdot 10^{-6}$
1,1,1-Trichloroethane	133.41	16,500	1,250	2.49	$7.9 \cdot 10^{-6}$
Chloroethylene	62.5	354,600	2,763	1.38	$1.3 \cdot 10^{-5}$ *2)
1,1-Dichloroethylene	96.94	80,500	3,344	2.13	$1.0 \cdot 10^{-5}$ *3)
cis-1,2-Dichloroethylene	96.94	27,000	3,500	1.86	$1.0 \cdot 10^{-5}$ *3)
trans-1,2-Dichlorethylene	96.94	44,400	6,260	1.93	$1.0 \cdot 10^{-5}$ *3)
Trichloroethylene	131.39	9,900	1,400	2.53	$8.8 \cdot 10^{-6}$
Tetrachloroethylene	165.83	2,415	240	2.88	$8.0 \cdot 10^{-6}$

*1) Estimated on the basis of the diffusion coefficient of dichloromethane and formula 1.

*2) Estimated on the basis of the diffusion coefficient of 1,1-dichloroethane and formula 1.

*3) Estimated on the basis of the diffusion coefficient of trichloroethylene and formula 1.

Table 5
Chemical data for phenols.

Substance name	Molecular weight	Vapour pressure	Water solubility	Octanolwater distribution coefficient	Acid dissociation constant	Diffusion coefficient in air
	m	p	S	log K_{ow}	pK_a	D_L
	g/mol	Pa	mg/l	-	-	m^2/s
Phenol	94.1	26.7	84,000	1.5	10.0	$8.5 \cdot 10^{-6}$ *)
o-Cresol	108.1	32.0	24,500	2	10.3	$7.9 \cdot 10^{-6}$ *)
p-Cresol	108.1	14.7	23,000	2	10.3	$7.9 \cdot 10^{-6}$ *)
2,4-Dimethylphenol	122.2	13.1	4,200	2.4	10.6	$7.5 \cdot 10^{-6}$ *)
2-Chlorophenol	128.6	189.3	28,500	2.2	8.5	$7.3 \cdot 10^{-6}$ *)
2,4-Dichlorophenol	163.0	16.0	4,500	3.1	7.9	$6.5 \cdot 10^{-6}$ *)
2,4,5-Trichlorophenol	197.5	2.9	1,200	3.9	7.4	$5.9 \cdot 10^{-6}$ *)
2,4,6-Trichloropehnol	197.5	2.3	800	3.1	7.4	$5.9 \cdot 10^{-6}$ *)
Pentachlorophenol	266.3	0.019	14	5.0	4.7	$5.1 \cdot 10^{-6}$ *)

*) Estimated on the basis of the diffusion coefficient of benzene and formula 1.

References

/1/ Miljøstyrelsen. Projekt om jord og grundvand nr. 20, 1996*: Kemiske stoffers opførsel i jord og grundvand* (The Environmental Protection Agency. Project on soil and groundwater No. 20, 1996: *Chemical Substance Behaviour in Soil and Groundwater*')

/2/ Lugg, G.A. 1968: *Diffusion Coefficients of Some Organic and Other Vapor in Air*. Analytical Chemistry, 40, 1072-1077.

Appendix 5.6

Risk assessment of groundwater - formulae for calculations

This Appendix provides an introduction to mixing, phase transitions, and spreading processes as they may occur in soil. More detailed descriptions can be found in literature, /1, 2, 3, 4, 5/.

For the saturated zone, a three-stage risk assessment is described.

- Stage 1 is a mixing model close to the source, based on the conservative assumption that the soil water at the bottom of the unsaturated zone has a contaminant concentration equal to the concentration strength at source. Subsequently, mixing in the uppermost 0.25 m of the groundwater aquifer is assumed.

  Alternatively, the resultant contamination concentration of the uppermost 0.25 m of the unsaturated zone can be directly determined by analysing groundwater extracted from a screen, which has been installed at the top of the groundwater aquifer.

- Stage 2 is a mixing model downgradient from the source, where increasing depths of mixing are assumed due to dispersion effects.

- At Stage 3 the resultant contamination concentration of the groundwater is calculated while taking account of dispersion, sorption, and degradation in the saturated zone. Stage 3 is an extension of Stage 2, as the starting point for Stage 3 is the resultant contamination concentration calculated at Stage 2.

The descriptions of models for substance spreading include a series of calculation parameters, some of which could be taken as standard parameters from Appendix 5.8, tables, or textbooks. Calculation parameters that typically have to be found in tables are designated as standard parameters in the text.

Examples of the use of the formulae indicated in connection with actual risk assessments can be found in Appendix 5.7.

A simple risk assessment cannot be outlined for the unsaturated zone. However, for the sake of comprehensiveness, equations for calculating substance concentrations (one-dimensional substance transport) in the unsaturated zone are provided.

1 Mixing model close to the source

No account is taken for neither sorption, dispersion, degradation, nor diffusion. It is assumed that the groundwater aquifer is homogenous (a one-layer model), and that the groundwater moves at constant speed.

The following section features these designations:

N = net infiltration $[LT^{-1}]$
A = extent of the contaminated area $[L^{2}]$
B = contaminated area width $[L]$
C_o = concentration strength at source $[ML^{-3}]$
d_m = mixing thickness $[L]$
V_D = Darcy velocity of groundwater $[LT^{-1}]$
C_g = Natural background contaminant concentration of groundwater $[ML^{-3}]$
k = hydraulic conductivity $[LT^{-1}]$
i = hydraulic gradient [unitless]
V_p = average soil-water velocity for groundwater $[LT^{-1}]$

The water flux Q_o, percolating through the contaminated area can be described as follows:

Equation 1
$Q_o = N \cdot A,$

And the flux J_o of contaminants as:

$J_o = C_o \cdot Q_o = C_o \cdot N \cdot A$

When soil water infiltrates into the saturated zone, mixing occurs in the uppermost 0.25 m of the groundwater aquifer.

The groundwater flux Q_g of water flowing below the contaminated area, consequently being contaminated by percolating soil water, corresponds to the groundwater held in a box with a length of V_D (Darcy velocity of the water), a height of 0.25 m (mixing thickness) and a width of B (width of the contaminated area).

The groundwater flux Q_g flowing below the contaminated area is then:

Equation 3
$Q_g = B \cdot 0.25 \cdot V_D = B \cdot 0.25 \cdot k \cdot i,$

as $V_D = k \cdot i$, /1/.

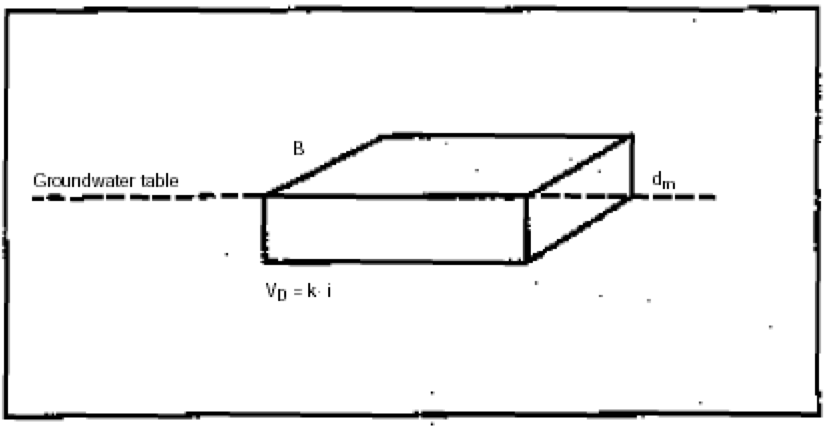

Figure 1
The flux of water below the contaminated area corresponds to the groundwater held in a box with a length of V_D (Darcy velocity of water), height of d_m (mixture thickness) and width B of the contaminated area.

When calculating the resultant contamination concentration of the groundwater, account must be taken of the fact that the groundwater may have a natural background concentration C_g of the contaminant.

For example, this applies to most metals.

The flux J_g of the natural contaminant content flowing with the groundwater below the contaminated area, can be expressed as:

Equation 4

$$J_g = Q_g \cdot C_g = C_g \cdot 0.25\ \mathrm{m} \cdot k \cdot i \cdot B$$

Where the background concentration of contaminants in the groundwater is caused by human activity at a different site, it should not be included in calculations. In these cases, J_g is set at 0.

The resultant contamination concentration C_1 in the groundwater flowing immediately below the contaminated area can then be expressed as the total sum of the contributions from the percolating soil water and from groundwater flowing in.

Equation 5

$$C_1 = \frac{J_o}{Q_o + Q_g} + \frac{J_g}{Q_o + Q_g}$$

Substituting the expressions from Equations 1-4 gives the following result:

Equation 6

$$C_1 = \frac{A \cdot N \cdot C_o + B \cdot 0.25\,\mathrm{m} \cdot k \cdot i \cdot C_g}{A \cdot N + B \cdot 0.25\,\mathrm{m} \cdot k \cdot i}$$

In the expression for calculating the resultant contamination concentration C_1, constant concentration at source C_o throughout the entire area is assumed.

If the investigation phase indicates that there is basis for such a procedure, the contaminated site can be divided into sections with individual concentrations at source; contamination concentrations weighted by area.

For contamination covering large areas, calculations may concentrate on the central area of contamination. This is in accordance with the principle that the zone with the highest concentration in the groundwater must comply with the groundwater-quality criteria.

The resultant contamination concentration in the uppermost 0.25 m of the unsaturated zone can also be determined directly by analysing groundwater extracted from a screen (with a screen length of 0.25 m) installed in the top of the groundwater aquifer. With regard to further risk assessment, the highest value of the concentrations measured is used.

Attention must be given to the fact that executing a well with a 0.25 screen requires accurate knowledge of the position of the groundwater table in order for the screen to be placed accurately. See Section 5.4 of these guidelines.

A screen with a greater effective screen length than 0.25 m can be used for measuring the resultant contamination concentration at the top of the groundwater aquifer, if sampling is conducted at very low pumping capacity, so that no significant depression cone appears.

In cases where a screen with a screen length greater than 0.25 m is used, the resultant contamination concentration C_1 in the uppermost 0.25 m of the aquifer must be calculated as follows:

Equation 7

$$C_1 = C_{1,measured} \cdot 1/0.25 \text{ m}$$

where $C_{1,measured}$ is the measured contamination concentration [ML^{-3}] and 1 is the effective screen length (in metres).

2 Downgradient mixing model

The soil water at the bottom of the unsaturated zone is conservatively assumed to have a contaminant concentration equal to the concentration at source. Subsequently, mixing in the uppermost 0.25 m of the groundwater aquifer is assumed.

The resultant contamination concentration is calculated at a point found at a distance from the contamination source corresponding to the groundwater’s transport distance per year (calculations are made using the soil water velocity of the groundwater), up to a maximum of 100 m. At this theoretical calculation point, all values must comply the groundwater criteria.

The assumptions are the same as for the mixing model close to the source. No account is taken of sorption, degradation, or diffusion. The groundwater aquifer is assumed to be homogenous and isotropic (a solo-stratum model), and the groundwater is assumed to maintain constant velocity.

The average soil-water velocity, which among other things is used when assessing the distance to the theoretical calculation point that must comply with the groundwater-quality criteria, is defined as follows:

Equation 8
$V_P = (k \cdot i)/e_{eff}$

where

k = hydraulic conductivity $[L\,T^{-1}]$
i = hydraulic gradient [unitless]

e_{eff} = effective porosity; a standard parameter found in Appendix 5.8.

Based on tests involving the spreading of radioactive tracers, an expression for the mixing depth d_m can be determined /4/:

$$d_m = 6\sqrt{2 \cdot D_{T,V} \cdot t}$$

where

$D_{T,V}$ is the downward dispersion coefficient $[L^2T^{-1}]$
T is the transport time of the groundwater [T].

Under Danish conditions, as established e.g. from studies at the landfill at Vejen /1/, it is deemed to be a conservative estimate to say that $D_{T,V} = 1/900\ D_L$, where D_L is the longitudinal dispersion coefficient. As $D_L = \alpha_L \cdot V_p$ the result is:

$$d_m = 6\sqrt{\frac{2}{900} \cdot D_L \cdot t}$$

$$= \sqrt{\frac{72}{900} \cdot \alpha_L \cdot V_p \cdot t}$$

where

α_L is the longitudinal dispersivity [L]
V_p is the soil-water velocity $[L\,T^{-1}]$
T is the observed transport time [T].

If the aquifer thickness is less than d_m, the actual aquifer thickness must be used.

The longitudinal dispersivity varies with the distance from the source of contamination. Standard values are found in Appendix 5.8.

The transport time to the theoretical calculation point cannot exceed one year. The transport time to the theoretical calculation point will often be less than one year; this applies when the soil-water velocity V_p is greater than 100 m/year.

The designations used in the following are identical to those used at the first stage of the risk assessment.

The resultant contamination concentration C_2 in the groundwater is calculated in a completely analogous fashion to the calculations for the first stage of the risk assessment, the downgradient mixing model.

Equation 10

$$C_2 = \frac{A \cdot N \cdot C_o + B \cdot d_m \cdot k \cdot i \cdot C_g}{A \cdot N + B \cdot d_m \cdot k \cdot i}$$

$$\cong \frac{A \cdot N \cdot C_o + B \cdot d_m \cdot k \cdot i \cdot C_g}{B \cdot d_m \cdot k \cdot i}, \text{ for } A \cdot N \ll B \cdot d_m \cdot k \cdot i$$

The expression used for calculating the resultant contamination concentration C_2 assumes a constant source-strength concentration C_o throughout the entire contaminated area. As with Stage 1 of the risk assessment, the contaminated area can be divided into areas with separate concentrations at source, which corresponds to contamination concentration weighted by area.

In cases where the contamination concentration C_1 in the uppermost 0.25 m of the groundwater aquifer has been measured during Stage 1, and any relevant subsequent corrections due to screen length have been made, this value can be used for simple calculation of the contamination concentration at a distance from the source C_2.

At the relevant calculation point, the resultant contamination concentration C_2 can be expressed as:

Equation 11

$$C_2 = C_1 \cdot (0.25\text{ m}/d_m)$$

$$= C_{1,\text{measured}} \cdot (l/d_m)$$

where

$C_{1,\text{measured}}$ is the resultant contamination concentration $[ML^{-3}]$ in the uppermost 0.25 m of the groundwater zone at the source of contamination; d_m is the mixing thickness after one year of groundwater transport, at a maximum distance of 100 m downgradient of the contamination. If the mixing thickness is less than 0.25 m, $d_m = 0.25$ m.
l is the effective screen length (in metres).

3 Downgradient mixing with degradation

Stage 3 of the risk assessment is an extension of Stage 2, as the starting point from Stage 3 is the resultant contamination concentration C_2, which was calculated at Stage 2.

For this reason, the resultant contamination concentration is calculated, as was the case at Stage 2 of the risk assessment, at a point situated at a distance from the contamination source which corresponds to the transport distance per year of the groundwater (calculations are based on the soil-water velocity of the groundwater); up to a maximum distance of 100 m. At this theoretical calculation point, the groundwater-quality criteria must be complied with.

Whereas Stages 1 and 2 of the risk assessment are, as has been mentioned, conservative models, it is not possible to carry out strictly conservative calculations at Stage 3. For this reason, monitoring must be carried out where degradation is involved.

The saturated zone is assumed to be homogenous and isotropic and to have a constant groundwater velocity. Degradation and vertical dispersion is assumed to occur in the saturated zone.

It is assumed that the degradation can be described as a 1^{st} order degradation. Calculations are carried out based on typical 1^{st} order degradation constants, which are not necessarily conservative.

On the basis of 1^{st} order degradation, the resultant contamination concentration C_3 after degradation can be expressed as /1,5/:

$$C_3 = C_2 \cdot \exp(-k_1 \cdot t)$$

Where

C_2 is the resultant contamination concentration as calculated in the downgradient mixing model at Stage 2 $[ML^{-3}]$
k_1 is the 1st order degradation constant in the saturated zone $[T^{-1}]$
t is the period of degradation [T]

Typical 1st order degradation constants have only been compiled for the BTEXs, for some of the chlorinated solvents, and for phenol /5/. Degradation constants for these substances are found in Appendix 5.8.

Account can be taken of sorption in connection with an assessment of how long the contaminants are subjected to degradation. This is done by assuming that the contaminants move to the theoretical calculation point at a velocity V_s given by

Equation 13
$$V_s = V_p / R, \quad R > 1$$

where

V_p is the average soil-water velocity $[LT^{-1}]$
R is the retardation coefficient [unitless]

The retardation factor can be calculated on the basis of the distribution coefficient K_d, which is a function of the organic content in soil f_{oc} and of the octanol-water distribution coefficient K_{ow}. Values for K_{ow} are found in tables, see Appendix 5.8. Based on the assumption that $\log K_{ow} < 5$ and $f_{oc} > 0.1\%$, K_d can be calculated by means of Abdul’s formula /1/:

Equation 14
$$\log K_d = 1.04 \cdot \log K_{ow} + \log f_{oc} - 0.84$$

The retardation factor can then be calculated by means of this formula:

Equation 15
$R = 1 + \rho_b/e_w \cdot K_d$

where ρ_b is the bulk density of soil [ML^{-3}],
e_w is the soil's porosity when saturated with water [unitless],
and K_d is the distribution coefficient.

As mentioned in the above, Stage 3 of the risk assessment, where degradation is involved, is not strictly conservative. For this reason, monitoring must be carried out in cases where the risk assessment shows that natural degradation of contaminants in the groundwater means that the standards of the groundwater-quality criteria can be met. This monitoring checks that degradation occurs in accordance with the assumptions. In addition to this, redox conditions must be determined, and data must be obtained to serve as the basis for calculating the relevant 1st order degradation constant.

When the rate of degradation is to be determined, it is necessary make corrections to the measured contamination concentrations to account for the effects of sorption, dispersion, dilution, etc. (non-destructive processes). This can be done by comparing concentrations of the contaminant with concentrations of a non-degradable substance (a tracer), or by comparing a slowly degradable contaminant and a more rapidly degradable contaminant.

Once the corrected contamination concentrations have been calculated, the 1st order degradation constant can be determined graphically by means of a t log-linear plot of the standard contamination concentration as a function of time. The 1st order degradation constant is determined as the slope of the linear section of the plot /5/.

In cases where the concentrations are known as functions of distance, rather than as functions of time, these figures can be converted by using the actual transport velocities that were determined during the contamination investigation.

The following addresses how the influences of non-destructive processes as regards the contamination concentration can be taken into account. This is done on the assumptions that the flux from the source of contamination is approximately constant, and that the groundwater aquifer is homogenous.

Measurements of concentrations of tracers and contamination made in two or more wells along a line of flow can be use to make an estimate of the contamination concentrations these wells would have had, if degradation were the only process attenuating the contamination.

An ideal tracer is influenced by non-destructive attenuation to the same degree as the contaminant (the tracer has the same volatility and same sorption coefficient as the contaminant), and the ideal tracer is not influenced by degradation processes.

Based on the assumption that an ideal tracer is utilised, the following equation describes the downgradient contamination concentration, where degradation has been the only attenuation process between the points i and i-

1, placed downgradient along a line of flow (e.g. between the wells 3 and 2 in Figure 5.11) /5,6,7/:

Equation 16

$$C_{i,corr} = C_{i-1,corr}\left(\frac{C_i}{C_{i-1}}\right)\left(\frac{T_{i-1}}{T_i}\right)$$

where:

$C_{i,corr}$ = the corrected contamination concentration at point i.
$C_{i-1,corr}$ = the corrected contamination concentration at point i-1. (If i-1 is the first point (the point furthest upstream), $C_{i-1,corr}$ is set at the observed contamination concentration at this point).
C_i = the observed contamination concentration at point i.
C_{i-1} = the observed contamination concentration at point i-1.
T_i = the observed tracer concentration at point i.
T_{i-1} = the observed tracer concentration at point i-1.

On the basis of the contamination concentration at a measuring point positioned upstream, and on the basis of measurements of the relationship between concentrations of contaminants and tracers, this equation can be used to estimate the theoretical contamination concentration, which would result from degradation alone.

Equation 16 is conservative insofar as $C_{i,corr}$ will be greater than C_i if the tracer is degraded. This will lead to the rate of degradation being estimated at lower values.

If the contamination concentration is corrected solely on the basis of tracer concentration between two points (A and B), Equation 16 is simplified:

Equation 17

$$C_{B,corr} = C_B\left(\frac{T_A}{T_B}\right)$$

A convenient method for estimating degradation constants is to utilise a non-degradable contaminant as a tracer. One such contaminant could be trimethylbenzene (TMB), which exists in three isomeric states (1,2,3-TMB; 1,2,4-TMB; and 1,3,5-TMB) generally present in fuels in sufficient quantities (3-7%) to be detectable in the groundwater /8,9/.

TMB is recalcitrant under anaerobic conditions, but is relatively easily degradable under aerobic conditions. TMBs degree of persistence is site-specific, and the usefulness of this contaminant as a tracer must be assessed in each individual case.

Tetramethylbenzene is another contaminant that can potentially be used as a tracer in connection with fuel contamination. However, tetramethylbenzene often appears in such small quantities that it is difficult to detect.

An ideal tracer has the same volatility and sorption coefficient as the contaminant under investigation, However, TMB has a greater sorption coefficient (and consequently a greater retardation coefficient) than BTEXs. For this reason, TMB is transported through the groundwater zone at lower velocities than the BTEXs.

Consequently, with a tracer like TMB it is necessary to modify equation 16 to take these differences of velocity into account. However, this modification is not necessary in stationary constant conditions (i.e. where the flux of the contaminant and tracer is constant at each point of measurement).

When a tracer is transported at a velocity that is significantly slower than the rate of spreading of the contaminant being investigated, concentrations of contamination and tracers must be assessed according to identical transport times instead of identical transport distances in order to take into account the degradation of the tracer as well as of the contaminant. The relationship between the velocities of tracer and contaminant can be expressed as:

Equation 18

$$\frac{V_t}{V_S} = \left(\frac{V_P}{R_t}\right) / \left(\frac{V_P}{R_c}\right) = \frac{R_c}{R_t}$$

where:

V_t = tracer transport velocity.
V_S = contaminant transport velocity.
V_P = groundwater velocity.
R_t = tracer retardation coefficient.
R_c = contaminant retardation coefficient.

The fraction of the tracer disappearing during the period of the contaminant moving from point i-1 to point i is represented by the expression $R_c/R_t(1-T_i/T_{i-1})$. That is to say that the remaining tracer fraction will be $1-R_c/R_t(1-T_i/T_{i-1})$.

In cases where the sorption of the tracer varies clearly from the sorption of the contaminant being investigated, the corrected contamination concentration at point i can be represented by the following equation:

Equation 19

$$C_{i,corr} = C_{i-1,corr}\left(\frac{C_i}{C_{i-1}}\right)\left(\frac{1}{\left(1-\frac{R_c}{R_t}\left(1-\frac{T_i}{T_{i-1}}\right)\right)}\right)$$

Where:

$C_{i,corr}$ = the corrected contamination concentration at point i.
$C_{i-1,corr}$ = the corrected contamination concentration at point i-1. (If i-1is the first point (the point furthest upstream), $C_{i-1,corr}$ is set at the observed contamination concentration at this point).
C_i = the observed contamination concentration at point i.

C_{i-1} = the observed contamination concentration at point i-1.
T_i = the observed tracer concentration at point i.
T_{i-1} = the observed tracer concentration at point i-1.

Note that when R_c equals R_t, Equation 19 equals Equation 16.

As mentioned before, Equation 16 is conservative insofar as any degradation of the tracer will result in a low estimate of the rate of degradation. For this reason, with contamination with mixtures such as e.g. petrol, diesel, or similar substances, it will be possible to use one of the most persistent contaminants as a tracer, as low degradation of this contaminant will only lead to a more conservative estimate of the rate of degradation for the other contaminants.

4 Spreading in the unsaturated zone

Flow in the unsaturated zone of the soil is governed by gravitation and by capillary forces (differences in tension) dependant on soil-water content and on soil characteristics such as texture and soil-size distribution.

The following features equations for calculating contaminant concentration (one-dimensional contaminant transport) and substance-front velocity. Only contaminant transport with soil water is considered, and diffusion to air is discounted.
Actual cases of pulse addition are rare, and pulse additions to the unsaturated zone must often be seen as sources of continuous contamination of the saturated groundwater zone. For example, a broken oil tank can be said to give off pulse additions to the unsaturated zone. However, since the depth of the groundwater table causes a long transport period before the contamination front reaches the saturated zone, and since the dissolution of oil contaminants in infiltrating soil water is also a slow process, oil from a leaking oil tank will typically seep down to the groundwater zone over a period of decades. For this reason, seepage of oil components from a leaking oil tank to the saturated groundwater zone must usually be considered as continuous contamination.

With pulse addition of substances, the substance concentration in the unsaturated zone as a function of infiltration depth (z) and time (t), can be described by means of a one-dimensional contaminant transport equation /10/:

Equation 20

$$C(z,t) = \frac{M}{e_w \sqrt{4\pi \cdot D_L \cdot t}} \exp\left(- \frac{(z - V_p \cdot t)}{4 D_L \cdot t} \right)$$

where z is the depth of the calculation point [L],
e_w is the water content of soil [unitless],
M is substance addition [M/L^2],
D_L is the longitudinal dispersion coefficient in the unsaturated zone[L^2T^{-1}],
t is the time elapsed from the onset of contamination [T],
V_p is the pore velocity of water [LT^{-1}].

The dispersion coefficient expresses the substance dispersion due to variations of flow velocity in each individual pore, mixing due to variations in pore size along the transport route, and molecular diffusion. Transversal dispersion is discounted.

The longitudinal dispersion coefficient D_L can be determined by means of the soil-water velocity V_P and the longitudinal dispersivity α_L as follows:

$$D_L = \alpha_L \cdot V_P$$

Standard values for the dispersivity α_L are found in Appendix 5.8.

By continuous substance addition (one-dimensional substance transport), the concentration as a function of infiltration depth and time can be described as follows, /1,10/:

Equation 21

$$C(z,t) = \frac{C_o}{2}\left[\operatorname{erfc}\left(\frac{z - V_p \cdot t}{2\sqrt{D_L \cdot t}}\right) + \exp\left(\frac{V_p \cdot z}{D_L}\right)\operatorname{erfc}\left(\frac{z + V_p \cdot t}{2\sqrt{D_L \cdot t}}\right)\right]$$

where z is the depth of the calculation point [L],
C_0 is the concentration at source [ML^{-3}],
D_L is the longitudinal dispersion coefficient (standard data) [L^2T^{-1}],
t is the time elapsed from the onset of contamination [T],
V_p is the pore velocity of the water [LT^{-1}].

Transversal dispersion is also discounted in this case.

The solution for the substance transport equation is approximate, and is only defined for $t < V_p$ x z, where V_p x z is precisely the time lapse before the soil water reaches the saturated zone in cases where simple piston flow is assumed (no dispersion, sorption, or degradation).

Erfc is the complementary error function erfc(y) = 1 - erf(y), where erf(y) is defined as:

Equation 22

$$\operatorname{erf} y = \frac{2}{\sqrt{\pi}}\int_0^y e^{-x^2}\,dx$$

The substance-transport equations provided take no account of sorption. Modifications to the equations for substance transport are subsequently addressed to facilitate inclusion of the effects of sorption in calculations.

When calculating substance concentrations as functions of depth and time, the water's pore velocity V_p is used for substances without the property of sorption. For substances with the property of sorption, this pore velocity is substituted by the propagation velocity of the substance front V_s, which can be described as follows:

Equation 23

$$V_S = \frac{V_P}{R_u} = \frac{N}{e_w \cdot R_u}, \text{ idet } V_P = \frac{N}{e_w}$$

where

V_p is the pore velocity of water [$L T^{-1}$],
N is the net infiltration [$L T^{-1}$],
e_w is the water-saturated porosity [unitless],
R_u is the retardation factor in the unsaturated zone [unitless].

See explanatory notes for Equation 13.

References

/1/ *Kemiske stoffers opførsel i jord og grundvand* ('Chemical Substance Behaviour in Soil and Groundwater').
Project on soil and groundwater No. 20.
The Environmental Protection Agency, 1996.

/2/ Bear, Jacob:
Hydraulics of Groundwater.
McGraw – Hill Inc., 1979.

/3/ *Grundvandskontrol ved kontrollerede affaldsdeponier* ('Groundwater Monitoring by Controlled Waste Deposits).
Dakofa, 1985.

/4/ Freeze, F og Cherry, J. A:
Groundwater.
Prentice – Hall, Inc., 1979.

/5/ Kjærgaard, M., Ringsted, J.P., Albrechtsen, H.J. og Bjerg, P.L. 1998. *Naturlig nedbrydning af miljøfremmede stoffer i jord og grundvand* ('Natural Degradation of Alien Substances in Soil and Groundwater'). Report prepared by the Danish Geotechnical Institute and the Department of Environmental Technololgy (Technical University of Denmark) for the Environmental Protection Agency.

/6/ Wiedemeier, T.H. et al. 1995. *Technical Protocol for Implementing Intrinsic Remediation with Long-Term Monitoring for Natural Attenuation of Fuel Contamination Dissolved in Groundwater.* Air Force Centre for Environmental Excellence, Technology Transfer Division, Brooks Air Force Base, San Antonio, Texas.

/7/ Wiedemeier, T.H. et al. 1996. *Technical protocol for evaluating natural attenuation of chlorinated solvents in groundwater. Draft – Revision 1.* Air Force Centre for Environmental Excellence, Technology Transfer Division, Brooks Air Force Base, San Antonio, Texas.

/8/ Christensen, L.B., Arvin, E. og Jensen, B. 1987. *Olieprodukters opløselighed i grundvand* ('Groundwater Solubility of Oil Products'). Report for the Environmental Protection Agency.
Laboratory of Technical Hygiene, Technical University of Denmark.

/9/ Verschueren, K. 1983. *Handbook of Environmental Data on Organic Chemicals.* 2. ed. Van Nostrand Reinhold.

/10/ Høgh Jensen, K. og Storm, B. *Stoftransport i umættet og mættet zone – simple beregningsmetoder* ('Substance Transport in Unsaturated and Saturated Zones – Simple Methods of Calculation'). Stads- og havneingeniøren 2, 1985.

Examples of specific risk assessments of groundwater

This appendix features examples of how specific risk assessments have been carried out. The risk assessments are done in accordance with the principles indicated in Section 5.4 of these guidelines. The formulae applied in some of the calculations are described in Appendix 5.6. Standard calculation parameters can be found in Appendix 5.8, such as net precipitation, hydraulic conductivity, and 1st order degradation constants.

Example 1

Sand aquifer covered by clay
Turpentine contamination near the surface

Beneath thin layers of top soil and a layer of fill there is non-fissured clay to a depth of approximately 3.9 m below surface level. No signs have been found of any secondary aquifers in the clay. Beneath the clay is a meltwater sand aquifer to a depth of approximately 6.9 m below surface level.

The level of the groundwater table is at approximately 3.9 m below surface level, which broadly corresponds to the top of the sand stratum.

The sand stratum has a substratum of sticky tertiary clay. Figure 1 shows a geological cross-section of the site.

An enterprise producing wood preservatives operated on the site. Leakage from a faulty processing plant has caused turpentine contamination of the soil. This contamination has been found to be limited to a depth between 2.5 and 3 m below surface level.

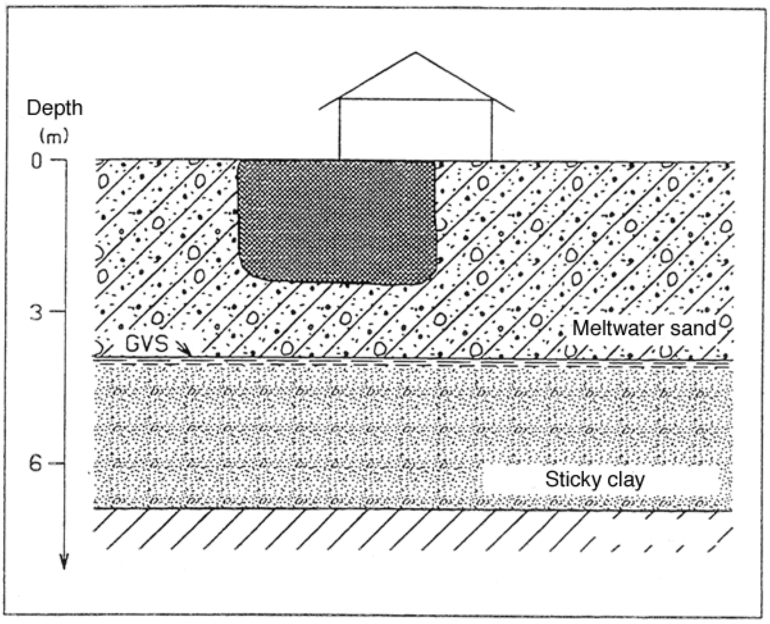

Figure 1
Cross-section showing geological conditions of example 1.

The contamination has spread all the way to the edge of existing buildings. Due to the very high cost of securing these building during the removal of soil, the contamination will only be dug away to a depth of 1.0 m below surface level

Soil samples from soil borings show concentrations of hydrocarbons of between 50 mg/kg and 800 mg/kg (quantified as turpentine) at depths of 1.0 to 3.0 m below surface level. This includes concentrations of single components up to: benzene 1.0 mg/kg, toluene 5mg/kg, and xylenes 12 mg/kg.

Figure 2 shows a map of benzene concentrations in the ground at 1.0 to 3.0 m below surface level. The concentrations indicated are maximum concentrations (mg/kg dry matter).

Present groundwater flow is generally south to south-west. Based on the assumption that the zone containing the highest concentration of contaminants in the groundwater must comply with the criteria for groundwater quality, the following risk assessment is carried out solely for the area that has been most affected by contamination (the area with a maximum benzene concentration of 1.0 mg/kg dry matter). This covers an area A of approximately 15 m x 8 m = 120 m^2.

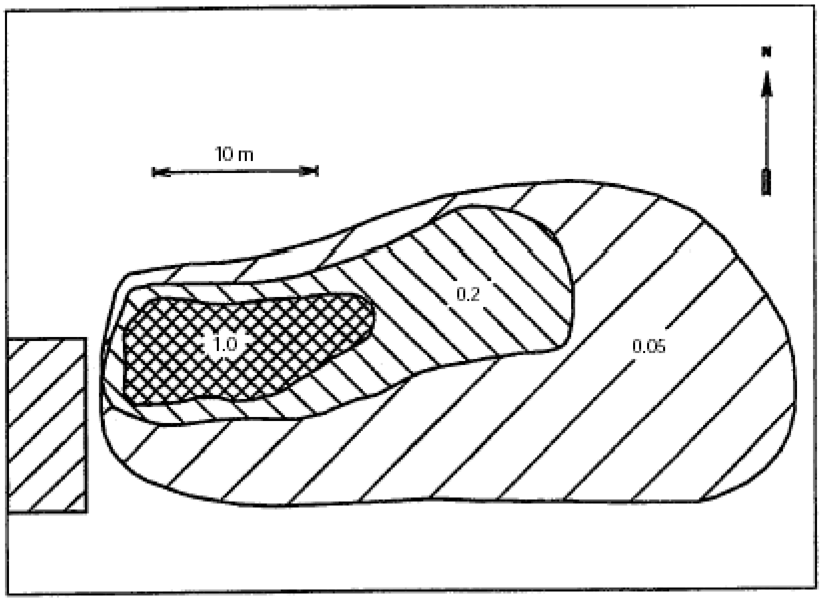

Figure 2
Example 1. Maximum benzene concentrations (mg/kg dry matter) 1.0 to 3.0 m below surface level.

Based on the surveys performed, the residual contamination of the area, after the removal of contaminated soil to a depth of 1.0 m below surface level, is estimated to comprise a maximum of 350 kg of turpentine (120 m^2 x 2.0 m x 1,800 kg/m^3 x 800 mg/kg), of this a maximum of 0.44 kg will be benzene.

Net precipitation in the region is approximately 240 mm a year (cf. Appendix 5.8). However, the groundwater division of the county authorities has carried out detailed groundwater models for the contamination area, and groundwater generation (net infiltration) of the site concerned is estimated at N = 100 mm/year. This means that percolation through the contaminated area is Q_0 = A (the contaminated area) x N (net infiltration) = 120 m^2 x 100 mm/year = 12 m^3/year.

Had this detailed knowledge of groundwater generation not been available, it would have been necessary to utilise the figures for net precipitation when calculating percolation.

Based on the specific capacity of the water abstraction soil borings nearby, the transmissivity of the groundwater aquifer is estimated at T= 8 x 10^{-4} m^2/s. With a saturated layer thickness of 3 m, this corresponds to a hydraulic conductivity of k = 2.7 x 10^{-4} m/s. The county authorities' contour map of groundwater levels shows that the hydraulic gradient of the groundwater aquifer is i=0.004 in the direction towards the water works.

The pore velocity of the water is V_p = (k x i)/e_{eff} = (2.7 x 10^{-4} m/s x 0.004)0.30, where e_{eff} is the porosity of the groundwater aquifer, which in this case (a sand aquifer) has been set at e_{eff} = 0.30 (cf. Appendix 5.8 – Table 1).

Dissolved oil products will be carried with percolating water through the unsaturated zone to the groundwater aquifer and onwards from here in the direction of the groundwater flow, i.e. in the direction of the water works' abstraction wells.

Both the saturated and the unsaturated zones are aerobic, which indicates a potential for degradation.

This means that the relevant processes are solution, transportation, sorption, dispersion, and degradation in the saturated as well as the unsaturated zones.

Risk assessment

The following is a review of a step-by-step risk assessment carried out with a view to assessing the effect of the benzene contamination on the subjacent groundwater aquifer. The groundwater-quality criterion to be met in the groundwater aquifer is 1 µg/l.

Risk assessment is carried out solely for the single component benzene, as the spreading of this single component (which has very great solubility, and for which a very low groundwater criterion exists) is estimated to be critical for the groundwater resource.

At Stage 1, the soil water at the bottom of the unsaturated zone is conservatively estimated to have a contaminant concentration equal to the concentration at source. Subsequent calculations assume simple mixing in the uppermost 0.25 m of the groundwater aquifer.

A conservative mixing model is also used at Stage 2. The resultant contamination concentration is calculated at a theoretical calculation point positioned at a distance from the contamination source corresponding to the transport velocity of groundwater over a one-year period, to a maximum of 100 m. Calculations use a mixing density d_m given by the equations in Appendix 5.6.

At Stage 3 of the risk assessment the resultant contamination concentration in the groundwater is calculated with consideration given to 1st order degradation. Stage 3 is an extension of Stage 2, as the point of departure is the contamination concentration determined at Stage 2.

Stage 1. Mixing model close to the source

Due to the difficulty of taking relevant water samples to determine concentrations at source, it has been decided to base the first stage of the risk assessment on simple considerations of solution and equilibrium.

The concentration at source has been calculated on the basis of an assumption of equilibrium between the phase distributions in water, air, and soil (the principle of fugacity). The calculation is carried out in accordance with the guidelines in Appendix 5.9. The calculations show that at a concentration in soil of 1.0 mg/kg DW, the benzene content in the soil-water is 5.0 mg/l (soil-particle density 2.7 kg/l; soil density 1.8 kg/l; volumetric proportion: soil 60%, air 10%, and water 30%).

As an alternative to land use of the principle of fugacity, the concentration at source can be set to equal the maximum solubility of benzene in water. However, as we have relatively low benzene concentrations in the soil, and as benzene is highly soluble in water, such an estimate will lead to far too great concentrations at source.

Conservative calculations must be made for the spreading of contamination. For this reason, it is assumed that no degradation or sorption of the oil products occur. Moreover, calculations disregard dispersion, i.e. are made for vertical percolation to the groundwater aquifer.

Benzene emissions with the infiltrating soil water from the remaining contamination will be approximately $J_0 = C_0 \cdot Q_0 = 5.0$ mg/l · 12m^3/year = 60 g/year.

Benzene does not occur naturally in groundwater, and consequently the background concentration C_g in the groundwater is set at nil.

The resultant benzene concentration C_1 in the uppermost part of the groundwater aquifer can then be calculated as stated in Appendix 5.6 – Equation 6:

$$C_1 = \frac{A \cdot N \cdot C_o + B \cdot 0.25[m] \cdot k \cdot i \cdot C_g}{A \cdot N + B \cdot 0.25[m] \cdot k \ i}$$

Substituting the appropriate values results in:

$$C_1 = \frac{120 \ m^2 \cdot 100 \ \text{mm/year} \ \cdot 5.0 \ \text{mg/l}}{120 \ m^2 \cdot 100 \ \text{mm/year} \ + 15 \ m \cdot 0.25 \ m \cdot 2.7 \cdot 10^{-4} \ \text{m/s} \ \cdot \ 0.004}$$

$$= 0.4 \ \text{mg/l}$$

The quality criteria for groundwater for benzene is 1 µg/l; see Section 6.4 of these guidelines. On the basis of a risk assessment based on simple mixing and completely conservative estimates (no degradation, dispersion, or sorption), it follows that at the top of the groundwater aquifer, in the stationary situation, a benzene concentration must be expected which clearly exceeds the quality criterion for benzene.

At Stage 1 of the risk assessment, all parameters in the calculations are linear. That is to say that for each parameter, uncertainty has the same impact on calculations as for the other parameters – each parameter is equally important.

A simple assessment of the rate of leaching can be made by outlining a mass balance. The remaining contamination was assessed to consist of approximately 350 kg turpentine, of which 0.44 kg benzene. At the calculated concentration at source of 60 g/year for benzene, approximately 8 years will pass before the benzene contamination is removed. No assumptions of unrealistically rapid removal of the source of contamination have been made.

The risk assessment which has been carried out shows that the groundwater-quality criterion for benzene cannot be met in this situation.

The reason for this may be that the concentration at source C_0 has been set too high due to a lack of analysis results. An option to consider, then, is to obtain analysis results from the uppermost area of the saturated zone from soil borings positioned immediately downgradient of the contaminated area, with very short (e.g. 0.25 m) screens installed in the uppermost area of the saturated zone. The very short screens are to ensure that inappropriate mixing with groundwater does not occur. There is a risk of this occurring, as the mixing depth is very small close to the source of contamination.

Another reason for the fact that the risk assessment shows that the groundwater criteria are not met may be that calculations are in general too conservative, as no account has been taken of sorption, dispersion, and degradation.

Stage 2. Downgradient mixing model

Stage 1 of the risk assessment provided the soil-water velocity V_p = 112 m/year. The theoretical calculation point is positioned at a distance from the source of contamination which corresponds to the groundwater's transport distance in a year, up to a maximum of 100 m. Consequently, in this case the calculation point is positioned at the maximum distance of 100 m.

The mixing density d_m is determined by (Appendix 5.6 - Equation 8):

$$d_m = \sqrt{\frac{72}{900} \cdot \alpha_L \cdot V_p \cdot t}$$

The value for α_L at a distance of 100 m from the source of contamination is determined from a figure (Appendix 5.8, Figure 2) at α_L = 0.4 m.

The mixing time t = (100 m)/ V_p = 100 m/(112 m/year) = 326 days.

Substituting these values gives the following result:

$$d_m = \sqrt{\frac{72}{900} \cdot 0.4 \text{ m} \cdot 112 \text{ m/year} \cdot 326 \text{ days}}$$

$$= 1.8 \text{ m}$$

In accordance with Appendix 5.6 – Equation 9, the resultant contamination concentration C_2 at the theoretical calculation point can be expressed by:

$$C_2 = \frac{A \cdot N \cdot C_o + B \cdot d_m \cdot k \cdot i \cdot C_g}{A \cdot N + B \cdot d_m \cdot k \cdot i}$$

The result after substitution is:

$$C_2 = \frac{120 \text{ m}^2 \cdot 100 \text{ mm/year} \cdot 5.0 \text{ mg/l}}{120 \text{ m}^2 \cdot 100 \text{ mm/year} + 15 \text{ m} \cdot 1.8 \text{ m} \cdot 2.7\ 10^{-4} \text{ m/s} \cdot 0.004}$$

$$= 64\ \mu\text{g/l}$$

Using the downgradient mixing model also results in values exceeding the groundwater criterion of 1 µg/l.

A choice can now be made between various options: Implementing remedial measures, carrying out stage 3 of the risk assessment, or carrying out a risk assessment on the basis of new data; for example contamination concentrations measured in the uppermost area of the saturated zone.

Stage 3. Spreading model with dispersion, sorption, and degradation

The following describes stage 3 of the risk assessment, where biological degradation in the saturated zone is taken account of.

A prerequisite for utilising stage 3 of the risk assessment is sufficient knowledge of local geology and hydrogeology to facilitate optimum positioning (vertically as well as horizontally) of soil borings for sampling and monitoring in the contamination plume and directly downgradient of the contamination. The investigation phase must also have shown that the redox conditions provide opportunities for degradation of the contaminants in question.

For this reason, an additional contamination investigation has been carried out in order to obtain sufficient background knowledge. This investigation has provided detailed charts of the groundwater flow and redox conditions.

The additional investigation uncovered anaerobic conditions in the groundwater immediately below the spillage site. No oxygen content has been detected (< 0.1mg/l), Fe(III) contents are lower than in upstream soil borings, and methane has been generated.

However, oxygen contents increase rapidly both upstream, sideways, and downgradient of the contamination; 10-12 m downgradient, oxygen contents of more than 1 mg/l are found, i.e. aerobic conditions – see Section 5.4.

In the additional investigation, water samples have been taken from the uppermost part of the groundwater aquifer immediately below the spillage site. The effective screen length, l, was 0.75 m. The highest concentration of benzene was determined to be 6.4 µg/l.

According to Appendix 5.6, Equation 7, the resultant contamination concentration C_2 in the most contaminated area of the groundwater zone at a distance of 100 m (mixing depth d_m) can be expressed as:

$$\begin{aligned} C_2 &= C_1 \cdot (0.25/d_m) \\ &= 19.2\ \mu g/l\ (0.25/1.8) \\ &= 2.7\ \mu g/l \end{aligned}$$

where:

$$\begin{aligned} C_1 &= C_{1,measured} (l/0.25\ m) \\ &= 6.4\ \mu g/l\ (0.75/0.25) \\ &= 19\ \mu g/l \end{aligned}$$

The groundwater-quality criteria of 1 µg/l is still not met.

The fact that there are aerobic conditions in the groundwater aquifer indicates a potential for degradation. The Fe(III)-reducing and

methanogenic conditions around the spillage site indicate that natural degradation of the contamination is already occurring.
For this reason, it is relevant to include degradation in the calculations, and the resultant contamination concentration C_3 after degradation, 100 m downgradient of the contamination, can be expressed as (Appendix 5.6 – Equation 12):

$C_3 = C_2 \cdot \exp(-k_1 \cdot t)$

The 1st order degradation constant k_1 can (cf. Appendix 5.8) be set at 0.01-0.2 day^{-1}. A conservative choice would then be $k_1 = 0.01\ day^{-1}$.

When assessing the amount of time that contaminants are subjected to contamination, account may be taken of sorption. This is done by calculating the movement of contaminants towards the theoretical calculation point at the velocity V_s expressed by (Appendix 5.6 – Equation 13):

$V_s = V_p / R, \quad R > 1$

Where:

V_p is the average soil-water velocity $[LT^{-1}]$.
R is the retardation coefficient for benzene [unitless].

The retardation factor depends on the substances involved and on soil bulk density ρ_b, actual content of organic substances in soil f_{oc}, and on the octanol-water distribution coefficient K_{ow}. The contents of organic substances f_{oc} for various types of soil are found in Appendix 5.3, Table 1. K_{ow} values for various substances are found in Appendix 5.5, Tables 1-5.

If $\log K_{ow} < 5$ and $f_{oc} > 0.1\%$, the distribution coefficient K_d can be calculated by means of Abdul's formula:

$\text{Log } K_d = 1.04 \cdot \log K_{ow} + \log f_{oc} - 0.84$

The retardation factor can then be calculated by means of the formula (Appendix 5.6 – Equation 15):

$R = 1 + \rho_b/e_w \cdot K_d$

The following relevant values for benzene in a sand deposit are determined from tables (Appendix 5.5 – Table 1 and Appendix 5.3 – Table 1):

$F_{oc} = 0.002$
$\text{Log } K_{ow} = 2.1$
$E_w = 0.45$
$\rho_b = 1.8$

which upon substitution provide the retardation factor for benzene:

$R_{benz} = 1.2$
$V_{S\text{-}benz} = 93$ m/year

Of which $t = 100\ m/V_{s\text{-}benz} = 390$ days.

The resultant contamination concentration C_3 of benzene at the calculation point 100 m downgradient of the contamination is then:

$$\begin{aligned} C_3 &= C_2 \cdot \exp(-k_1 \cdot t) \\ &= 2.7\ \mu g/l \cdot \exp(-0.01 \cdot 390) \\ &= 0.1\ \mu g/l \end{aligned}$$

In this case, the groundwater quality criterion for benzene (1 µg/l) is clearly met.

However, it is not sufficient to have calculations show that the quality criterion for groundwater is met. It is necessary to carry out monitoring which shows that the degradation of contamination occurs as predicted, and that the redox conditions continue to provide opportunities for degradation. Moreover, the current degradation constant should be calculated.

A number of soil borings were carried out in connection with the additional investigation. Four of these soil borings (designated A-D) are positioned on a flow-line downgradient of the contamination at distances from 4 to 37 m.

Selected monitoring data are shown in Table 1. As is apparent, there are aerobic conditions in soil borings B, C, and D (oxygen contents are greater than 1 mg/l), whereas there are low-oxygen conditions (probably anaerobic conditions) in soil boring A. This means that redox conditions have not changed since the additional investigation was carried out, and consequently there is still a potential for degradation of the benzene contamination.

The contamination concentration in the monitoring soil borings is less than expected. To check these values, the current degradation constant is calculated in accordance with Appendix 5.6.

Corrections are made to take into account sorption, dispersion, diffusion, etc.

The corrected contamination concentrations are determined by (Appendix 5.6 – Equation 19):

$$C_{i,corr} = C_{i-1,corr}\left(\frac{C_i}{C_{i-1}}\right)\left(\frac{1}{\left(1-\frac{R_c}{R_t}\left(1-\frac{T_i}{T_{i-1}}\right)\right)}\right)$$

where:

$C_{i,corr}$ = corrected contamination concentration at point i.

$C_{i-1,cor}$ = corrected contamination concentration at point i-1.(Where point i-1 is the first point (positioned furthest upstream), $C_{i-1,corr}$ is set to equal the observed contamination concentration).

C_i = observed contamination concentration at point i.

C_{i-1} = observed contamination concentration at point i-1.

T_i = the observed tracer concentration at point i.

T_{i-1} = the observed tracer concentration at point i-1.

R_c = the retardation coefficient for the contaminant.

R_t = tracer retardation coefficient.

Xylene is used as tracer, as xylene is degraded at a considerably lower rate than benzene in this case. (see Table 1). The fact that xylene is degradable means that it is not an ideal tracer, but this results in a conservative estimate; degradation is underestimated.

Table 1.
Selected data from 4 monitoring wells positioned on a line downgradient of the contamination .

Monito-ring Well	Downgradient distance	Transport time benzene (days)	Xylene content (μg/l)	Benzene content (μg/l)	Corrected benzene content (μg/l)	Oxygen content (mg/l)
A	4 m	0	12.0	6.4	6.4	0.3
B	15 m	43	9.2	2.4	2.7	1.7
C	25 m	82	6.5	0.7	0.9	1.8
D	37 m	145	5.0	0.09	0.13	2.1

Input data comprises values from Table 1and the previously calculated retardation coefficient for benzene (R_{Benz} = 1.2). Also used is the retardation coefficient for xylene R_{xyl}, which is calculated correspondingly; R_{xyl} = 2.9.

To serve as an example, we calculate the contamination concentration in soil boring B with xylene corrections. Substitution results in:

$$C_{B,corr.} = C_{A,corr.} \left(\frac{C_B}{C_A}\right) \left(\frac{1}{1-\frac{R_{Benz}}{R_{xyl}}\left(1-\frac{T_B}{T_A}\right)}\right)$$

$$= 6{,}4 \left(\frac{2{,}4}{6{,}4}\right) \left(\frac{1}{1-\frac{1{,}2}{2{,}9}\left(1-\frac{9{,}2}{12{,}0}\right)}\right) \text{mg}/\text{l}$$

$$= 2{,}7\,\text{mg}/\text{l}$$

Benzene concentrations with xylene corrections are included in Table 1.

The corrected concentrations are then plotted into a log – linear plot as a function of the transport period from soil boring A (see Figure 1), and the 1st order degradation constant k_1 is determined as the slope of the linear section of the curve.

Figure 3 shows how these points are really positioned on a straight line. This means that the requirement that degradation should be able to be designated as a 1st order degradation is met.

In this case, benzene concentrations drop from approximately 7.0 µg/l on day 0 to approximately 0.1µg/l after 150 days. The slope of the curve in Figure 3, and consequently the actual degradation constant, can then be determined as:

$-k_1 = \ln(C/C_0)/t = \ln(0.1/7.0)/(150\text{days}) = -0.03 \text{ days}^{-1}$.

This is to say that at the time in question, the actual degradation constant is greater than the conservative value used in the risk assessment. We are on the safe side.

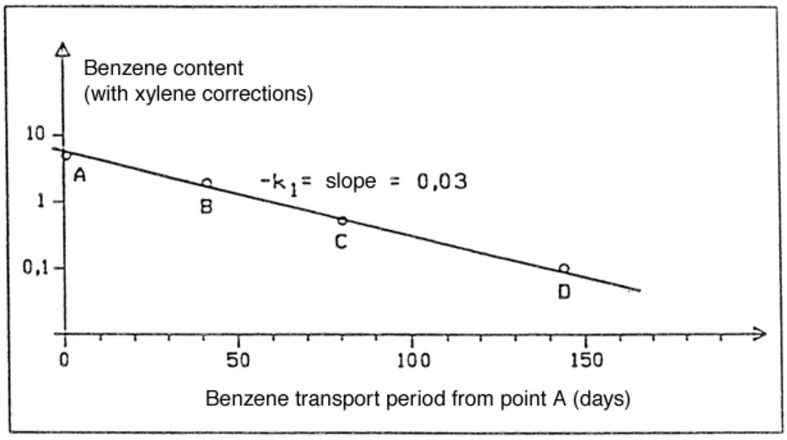

Figure 3
Plot of benzene content with xylene corrections along the flow line A, B, C. D.

Monitoring must be repeated to ensure that no changes occur, for example to the redox conditions, to stop or severely curtail degradation. As a minimum, monitoring must typically take place twice a year for three years.

Example 2

Sand aquifer covered by sandy clayey till
TCE contamination (Trichloroethylene)

A layer of fill of a 1.0 m thickness is uppermost. Below this are 2.0 metres of coarse sedimentary sand with a substratum of clayey till to a depth of approximately 8 m below surface level. The clayey till has a substratum of approximately 5 metres of sand and gravel. Beneath the lower sand stratum is sticky tertiary clay.

Figure 4 shows a geological section of the site.

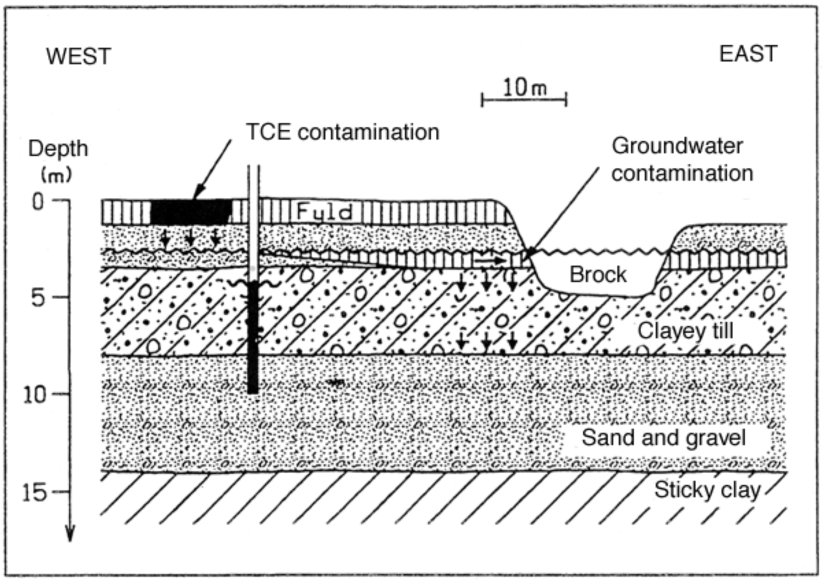

Figure 4
Section showing the geological conditions of example 2

There is a upper as well as a lower groundwater aquifer.
The upper aquifer is situated in the upper sand stratum.
The aquifer is unconfined with a water table level situated approximately 2.5 metres below surface level.

A brook flows approximately 40 metres east of the survey area.
The groundwater flow in the upper aquifer flows towards this brook with a hydraulic gradient i = 0.006; as established from observation of water levels in wells. Net precipitation N is estimated at 200 mm a year (cf. Appendix 5.8). On this basis infiltration through the contaminated area can be established as Q_o= A (the contaminated area) x N (net precipitation) = 60 m^2 x 200 mm/year = 12 m^3 /year.

County groundwater authorities estimate the hydraulic conductivity k of the upper aquifer at k = 2.5×10^{-4} m/s.

The average pore water velocity V_p in the upper groundwater aquifer can then be set at (cf. Section 5.6 – equation 8):

$$\begin{aligned} V_p &= (k \times i)/e_{eff} \\ &= (2.5 \times 10^{-4} \text{ m/s} \times 0.006)/0.30 \\ &= 157 \text{ m/year} \end{aligned}$$

where e_{eff} is the effective porosity of the groundwater aquifer, which in this case has been set at 0.30 (cf. Appendix 5.8 – Table 1).

The lower sand stratum constitutes the lower groundwater aquifer. The aquifer has a head approximately 3.5 m below surface level. That is to say that there is a pressure difference of approximately a 1.0 m water column between the lower and the upper aquifer, and a downward gradient.

On the basis of test pumping, hydraulic conductivity k of the lower groundwater aquifer is estimated at : $k = 3.6 \times 10^{-4}$ m/s. The hydraulic gradient is determined as: $i = 0.005$; as with the upper aquifer, the groundwater flow runs broadly to the east.

The average pore water velocity V_p of the water of the lower groundwater aquifer can now be determined as (cf. Appendix 5.6 – equation 8):

$$
\begin{aligned}
V_p &= (k \times i)/e_{eff} \\
&= (3.6 \times 10^{-4} \text{ m/s} \times 0.005)/0.30 \\
&= 190 \text{ m/year}
\end{aligned}
$$

where the effective porosity $e_{eff} = 30$ % has been established previously in connection with a nearby abandoned water work well.

It has been established that the layer of fill contains trichloroethylene (TCE) in an area A covering approximately 50 m^2 (approximately 12 m x 4 m) and throughout the entire depth of the layer of fill (i.e. 1.0 m). TCE concentrations are between 4 to 6 mg/kg dry matter, i.e. almost evenly distributed. The soil quality criteria for TCE is 5 mg/kg dry matter (Section 6.2 of these guidelines), so the concentrations found were actually similar to these soil-quality criteria.

However, since the test area has been designated as a car park, the area is not deemed to pose any risk to this form of land use.

No TCE has been detected in the underlying intact soil layer.

On the basis of the surveys performed, contamination is estimated at a maximum of $50 m^2 \times 1 \text{ m} \times 1.7 \text{ tonnes}/m^3 \times 6 \text{ mg/kg} = 0.5$ kg.

The concentration at source C_0 can be calculated on the assumption of equilibrium exists between the phase distributions in soil, air, and water (the principle of fugacity). This calculation, made in accordance with the directions found in Appendix 5.9, show that at a TCE concentration in soil of 6 mg/kg dry matter, the TCE content of pore water (concentration at source C_0) is approximately 8 mg/l (the particle density of the soil is 2.65 kg/l; the density of the soil is 1.7 kg/l; volumetric proportions: soil 55%, air 30%, and water 15%).

The only nearby recipient in the direction of the flow is a brook approximately 40 m downgradient of the contaminated area.

The upper aquifer is of no interest with regard to water abstraction. For this reason, risk assessment will only address contamination hazards to the lower aquifer and the nearby brook.

The width of the vertical dispersion zone d_m of the upper groundwater aquifer can be determined as (Appendix 5.6 – equation 9):

$$d_m = \sqrt{\frac{72}{900} \cdot \alpha_L \cdot V_p \cdot t}$$

At the brook (a distance of approximately 40 m from the contamination) the following vertical width is found:

$$d_m = \sqrt{\frac{72}{900} \cdot 0.15\ m \cdot 190\ m\ /\ year\ \cdot \frac{190\ m\ /\ year}{40\ m}} \ = \ 0{,}7\ m$$

where α_L = 0.15 has been read off from Figure 2 in Appendix 5.8, and time of flow t= 190 m/year/40 m.

As the upper aquifer is only 0.5 m thick, this means that the contaminants are mixed throughout the entire thickness of the aquifer before the groundwater reaches the brook.

Similarly, the zone of dispersion at a distance of 30 m downgradient of the contamination can be calculated as:

$$d_m = \sqrt{\frac{72}{900} \cdot 0.10\, m \cdot 30\, m} \ = 0.5\, m$$

In terms of these calculations, we have reached mixture of the contaminants throughout the upper aquifer 30 m downgradient of the contamination.

This means that the leach area, defined as the area where leaching from the upper aquifer through the clayey till to the lower groundwater aquifer takes place, extends from 30 to 40 m downgradient of the contamination, since the upper aquifer is cut off by the brook 40 m downgradient of the contamination.

The resultant contamination concentration C_s of the upper groundwater aquifer at a distance of 40 m from the contamination source (i.e. just as the groundwater reaches the brook) can be calculated as (Appendix 5.6 – equation 10):

$$Cs = \frac{A \cdot N \cdot C_o + B \cdot d_m \cdot k \cdot i \cdot C_g}{A \cdot N + B \cdot d_m \cdot k \cdot i}$$

TCE does not occur naturally in groundwater; hence the background concentration C_g of the groundwater is zero.

Substituting into the equation gives :

$$C_S = \frac{50\ m^2 \cdot 200\ mm\ /\ year \cdot 8\ mg/l}{50\ m^2 \cdot 200\ mm\ /year + 12\ m \cdot 0.5\ m \cdot 2.5 \cdot 10^{-4}\ m\ /\ s \cdot 0.006}$$

= 270 µg/l

A simple estimate of the rate of leaching can be made by a mass balance consideration. Residual contamination was estimated to comprise approximately 0.5 kg TCE. With a source strength J of J = N x A x C_0 = 50 m^2 x 200 mm/year x 8 mg/l = 80 g/year it will take approximately seven years to leach the TCE contamination. This means that the speed at which the source of contamination is removed is not unrealistic.

The above calculations show that the groundwater flowing from the upper aquifer into the brook has a TCE content of 270 µg/l.

We have also calculated that each year the contamination leaches an amount of TCE determined as:

$$\begin{aligned} A \text{ x } N \text{ x } C_0 &= 50 \text{ m}^2 \text{ x } 200 \text{ mm/year x } 8 \text{ mg/l} \\ &= 80 \text{ g/year} \end{aligned}$$

County authorities' recipient division informs us that the brook's flow of water Q_{min} at median minimum is approximately 2 l/second. In the brook, that has a flow of water corresponding to median minimum, the contaminated groundwater is diluted due to mixing. This gives a TCE concentration C_{brook} of:

$$\begin{aligned} C_{brook} &= A \text{ x } N \text{ x } C_0/Q_{min} \\ &= 80 \text{ (g/year)/ } 2 \text{ (l/second)} \\ &= 1.27 \text{ x } 10^{-3} \text{ µg/l} \end{aligned}$$

On the basis of the calculations performed, the county recipient division's assessment is that the TCE leaching into the brook does not pose any risk in relation to the objectives for the recipient.

The vertical velocity (Darcy velocity) V_D of the groundwater can be determined as (cf. Appendix 5.6 – equation 3):

$$V_D = k \text{ x } i$$

As with horizontal flow, the hydraulic gradient is expressed as the difference in water level Δh (in this case between the two aquifers) divided by the difference in distance Δs (in this case the vertical distance). We have:

$$\begin{aligned} I &= \Delta h/\Delta s \\ &= (3 \text{ m} - 2 \text{ m})/(8\text{m} - 3 \text{ m}) \\ &= 0.2 \end{aligned}$$

The vertical hydraulic conductivity can be established by looking it up in Appendix 5.8, Table 1. The lower boundary of the clayey till stratum determines hydraulic conductivity. In this case, the clayey till stratum goes down to a depth of 8 m below surface level.

By interpolating the values of the table, k is determined as 1 x 10^{-8} m/second.

This means that the vertical Darcy velocity is

$$\begin{aligned} V_D &= 0.2 \text{ x } 10^{-8} \text{ m/second} \\ &= 60 \text{ mm/year} \end{aligned}$$

Seen in relation to the fact that net precipitation is 200 mm/year, vertical velocity in the clayey till seems quite high, but still possible.

The flux J of TCE through the clayey till can be calculated as (Appendix 5.6 – equation 1):

J = V_D x A x Cs
= 60 mm/year x 10 m x 12 m x 270 µg/l
= 1.9 g/year

This flux through the clayey till does not seem unrealistically large when compared to the flux down towards the upper groundwater aquifer. It should, however, be noted that using table values for vertical hydraulic conductivity creates scope for considerable uncertainty.

For instance, if we take the example above and substitute a vertical hydraulic conductivity of e.g. 10^{-7} m/s (instead of 10^{-8}), this will provide us with a vertical Darcy velocity of 600 mm/year. This is three times more than the net precipitation and hence totally unrealistic.

Analogously to Appendix 5.6, equation 6 (Step 1 of risk assessment), the contamination concentration C_{P1} in the upper 0.25 m of the lower groundwater aquifer immediately below the percolation area (which is 30 – 40 m downgradient of the contamination) can be determined as:

$$C_{P1} = \frac{A \cdot V_D \cdot C_S}{A \cdot V_D + B \cdot 0{,}25m \cdot k \cdot i}$$

Where

A = The percolation area (through the moraine clay); 120 m^2.
V_D= Vertical Darcy velocity 60 mm/year.
Cs = Contamination concentration of the upper groundwater aquifer (where the aquifer has full mixture); 270 µg/l.
B = Breadth of soil contamination; 12 m.
K = Horizontal hydraulic conductivity of the lower aquifer; 3.6 x 10^{-4} m/s.
i = The hydraulic gradient of the lower aquifer 0.005

If these values are substituted, the result is:

C_{P1} = 11.0 µg/l.

The groundwater quality criteria of 1µ g/l is exceeded immediately below the percolation area.

For this reason, we apply the second step of risk assessment to the lower groundwater aquifer.

At an earlier stage, we calculated the average pore water velocity in the lower aquifer V_P to be 190 m/year. In the second step of risk assessment we need to calculate the contamination concentration at a point downgradient of the contamination at a distance corresponding to a year of groundwater flow (to a maximum of 100 m). In this example, the theoretical point of calculation is 100 m downgradient of the contamination.

The percolation area, i.e. the area where the clayey till has been infiltrated by TCE contaminated groundwater, is 30-40 m downgradient of the contamination.

That is to say that the theoretical calculation point is only 60 m away from the leach area.

The vertical dispersion zone d_m can now be determined as (Appendix 5.6 – equation 9):

$$d_m = \sqrt{\frac{72}{900} \cdot \alpha_L \cdot V_P \cdot t}$$

where $\alpha_L = 0.21$ has been read off from Figure 2 in Appendix 5.8, and degradation time t = (60m/190m/year)).

According to Step 2 of risk assessment (Appendix 5.6 – equation 10), the contamination concentration C_{P2} of the lower groundwater aquifer 100 m downgradient of the contamination can be calculated as:

$$C_{p2} = \frac{A \cdot V_D \cdot C_S}{A \cdot V_D + B \cdot d_m \cdot k \cdot i}$$

If we substitute the values, the result is:

$$C_{P2} = \frac{120 \text{m}^2 \cdot 0.06 \text{ m/year} \cdot 270 \text{ µg/l}}{120 \text{m}^2 \cdot 0.06 \text{ m/year} + 12 \text{ m} \cdot 1 \text{ m} \cdot 3.6 \cdot 10^{-4} \text{ m/s} \cdot 0.005}$$

$$= 2.8 \text{ µg/l}$$

This means that upon completion of Step 2 of the risk assessment, the contamination is still deemed to present a hazard to groundwater resources.

In order to carry out Step 3 of risk assessment (see Section 5.4 of these guidelines), it is necessary to have the knowledge of geological and hydrogeological conditions required to facilitate optimum location of wells for surveying and monitoring. Complicated flow conditions around the brook prevent these requirements from being met. Due to this, Step 3 of the risk assessment process will not be carried out.

Example 3

Sand aquifer without cover
Arsenic contamination

Below a thin cover of top soil is meltwater sand to a depth of approximately 5.0 m below surface level. Underlying the sand layer is a layer of sticky clayey till.

Figure 5 shows a geological section of the site.

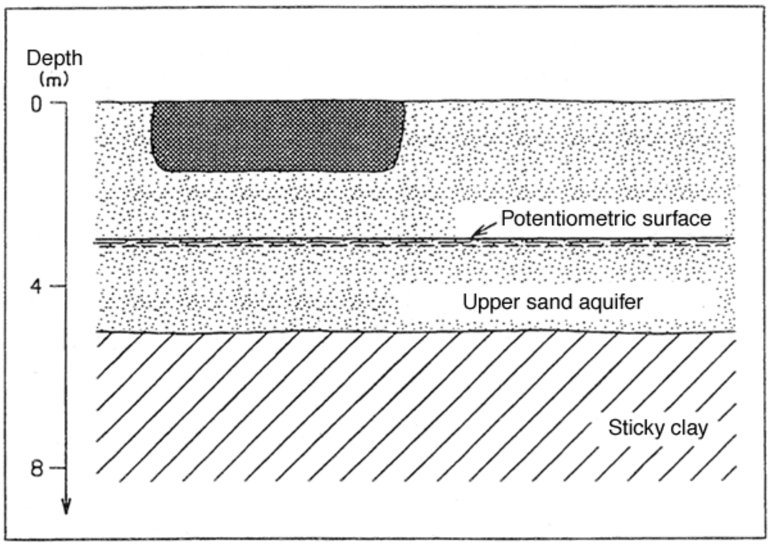

Figure 5
Section showing the geological conditions of example 3.

An area A of approximately 80 x 40 m has been contaminated by arsenic to a depth of approximately 1.5 m below surface level. The contamination was caused by leakage from a wood-preservation plant which was abandoned in the 1950s. The contaminated area is situated in a park-like area located close to built-up areas.

In general, the concentration of contamination in the area is around 50 mg/kg dry matter. There is, however, a hot spot where As concentrations of up to 1,200 mg/kg dry matter have been detected. This hot spot can be recognised from surface level to a depth of 0.5 m below surface level.

Since additional costs will be modest, the top 0.5 m of soil will be removed throughout the entire contaminated area. Re-establishing the area with uncontaminated soil will ensure that the top 0.5 m will be in accordance with the soil quality criteria of 20 mg/kg dry matter and the eco-toxicological soil quality criteria of 10 mg/kg dry matter (see Section 6.2 and 6.3 of these guidelines).

As regards usage of the site, the area is subject to administrative regulation preventing exposure to contaminants.

Total arsenic content of the soil is estimated at 3,200 m^2 x 1m x 50 mg/kg x 1.7 kg/m^3 = 270 kg.

Leaching surveys have previously been carried out from hot spots showing that pore water concentrations C_0 of up to 2 mg/l must be expected.

Groundwater monitoring shows that the natural arsenic content of the area’s groundwater is under 2 µg/l. The groundwater quality criteria for arsenic is 8 µg/l (see Section 6.5 of these guidelines).

The sand stratum constitutes an unconfined upper groundwater aquifer, and the groundwater table is situated approximately 3.0 m below surface level.

Downgradient of the survey area are several individual water-abstraction sites from the upper aquifer. This means that the aquifer must be protected as a groundwater resource.

On the basis of data from the abstraction wells, the hydraulic conductivity k of the aquifer is estimated at $k = 5 \times 10^{-4}$ m/s; the hydraulic gradient is estimated a 0.005, and effective porosity e_{eff} at 25 %.

Net precipitation N is set at 180 mm/year (cf. Appendix 5.8).

The groundwater's average pore water velocity V_p can be determined as (Appendix 5.6 – equation 8):

$$\begin{aligned} V_p &= (k \times i)/\ e_{eff} \\ &= (5 \times 10^{-4}\ \text{m/s} \times 0.005)/0.25 \\ &= 315\ \text{m/year} \end{aligned}$$

There are no downgradient recipients within the nearest 1,500 m.

A simple estimate of the rate of leaching can be made by a mass balance consideration. Residual contamination was estimated to comprise approximately 270 kg of arsenic. With a source strength J of $J = A \times N \times C_0 = 3{,}200\ \text{m}^2 \times 180\ \text{mm} \times 8\ \text{mg/l} = 1.15$ kg/year it will take slightly more than 200 years to leach the arsenic contamination. This does not seem unrealistic.

The contamination concentration C_1 in the uppermost 0.25 m of the aquifer immediately below the contamination can according to Appendix 5.6 – Equation 6 be calculated as:

$$C_1 = \frac{A \cdot N \cdot C_o + B \cdot 0.25\ \text{m} \cdot k \cdot i \cdot C_g}{A \cdot N + B \cdot 0.25\ \text{m} \cdot k \cdot i}$$

Substituting the appropriate values results in:

$$C_1 = \frac{3200\ \text{m}^2 \cdot 0.18\ \text{m/year} \cdot 2\ \text{mg/l} + 50\ \text{m} \cdot 0.25\ \text{m} \cdot 5 \cdot 10^{-4}\ \text{m/s} \cdot 0.005 \cdot 2\ \mu\text{g/l}}{3200\ \text{m}^2 \cdot 0.18\ \text{m/year} + 50\ \text{m} \cdot 0.25\ \text{m} \cdot 5 \cdot 10^{-4}\ \text{m/s} \cdot 0.005}$$

$$= 740\ \mu\text{g/l}$$

Where the width of the soil contamination B (50 m) is measured perpendicular to the direction of the groundwater flow.

This means that with the outlined assumptions, stage 1 of the risk assessment offers up resultant contamination concentration which are far greater than the groundwater quality criteria.

For this reason, stage 2 of the risk assessment is carried out.

After carrying out stage 1 of the risk assessment, a number of investigation soil borings with screen installations have been conducted in the groundwater aquifer.

The greatest arsenic concentration in the groundwater was found immediately below the soil contamination. An arsenic concentration C_1 of 45 µg/l was observed in a screen with an effective screen length l of 0.75 m.

In stage 2 of the risk assessment, the resultant contamination concentration is calculated for a theoretical calculation point which is situated at a distance from the contamination source corresponding to the annual soil-water transport distance (to a maximum of 100 m). In this case, a soil-water velocity V_P of 315 m/year was found. For this reason, the calculation point in this context is situated at the maximum distance of 100 m from the contamination source.

According to Appendix 5.6 – Equation 9, the mixing density d_m at a distance of 100 m can be determined as:

$$d_m = \sqrt{\frac{72}{900} \cdot \alpha_L \cdot V_P \cdot t}$$

$$= \sqrt{\frac{72}{900} \cdot 0.40\,\mathrm{m} \cdot 315\,\mathrm{m/year} \cdot 100\,\mathrm{m}/(315\,\mathrm{m/year})}$$

$$= 1.8\ \mathrm{m}$$

where α_L is found in Figure 2 in Appendix 5.8, and t = 100 m/(315 m/year).

The resultant contamination concentration C_2 in the theoretical calculation point positioned 100 m downgradient of the contamination can then be calculated as (cf. Appendix 5.6 – Equation 11):

$$\begin{aligned} C_2 &= C_1 \cdot (0.25\ \mathrm{m}/d_m) \\ &= 135\ \mu g/l \cdot (0.25/1{,}8) \\ &= 19\ \mu g/l \end{aligned}$$

where

$$\begin{aligned} C_1 &= C_{1,measured} \cdot (1/0.25\ \mathrm{m}) \\ &= 45\ \mu g/l \cdot (0.75/0.25) \\ &= 135\ \mu g/l \end{aligned}$$

So stage 2 of the risk assessment also comes up with resultant contamination concentrations greater than the groundwater criteria.

Arsenic is an element and as such cannot be degraded (only changes in its chemical form can occur). For this reason it is impossible to carry out stage 3 of the risk assessment, where degradation is included in calculations.

For this reason, remedial measures should be taken, as the contamination poses a threat to the groundwater resource.

Standard data to be used for risk assessments of groundwater

A series of calculation parameters must be used in connection with risk assessments of groundwater. This appendix provides examples of standard data which can be used in these calculations.

It must be stressed that more conservative values must be used if the parameters used are estimates, regional rather than local, or uncertain for other reasons. If risk assessment is to be precise and conservative estimates are to be avoided, more calculation data must be site-specific.

1 Groundwater recharge

Calculations of the amount of infiltrating soil water at a contaminated site include assessments of groundwater recharge proportion of precipitation.

In certain local areas, counties have knowledge of the proportion of precipitation which comprises groundwater recharge. However, for most areas it is necessary to use net precipitation as a conservative alternative. Figure 1 outlines the average net precipitation in Denmark and its distribution among areas /1/.

Evaporation from soil and plants (actual evaporation) is included in calculations of net precipitation and is established by means of model calculations by the computer model EVACROP. Evaporation varies with crop type/vegetation type and soil type. In these calculations, crops are estimated to comprise a mixture of 50% winter crops and 50% summer crops, a mixture which will cover 60-70% of the agricultural areas. The estimated soil type for Jutland is a mixture of coarse and fine grain soil, and the soil type for the rest of Denmark is set to be clay soil mixed with sand. These assumptions result in an actual evaporation of 400 mm per year for Jutland and 440 mm per year for the rest of Denmark. Local deviations from these typical values will occur due to land use and crop distribution. As net precipitation is the difference between precipitation and actual evaporation, this will occasion deviations from the values shown in Figure 1; for this reason, these values will have certain imperfections.

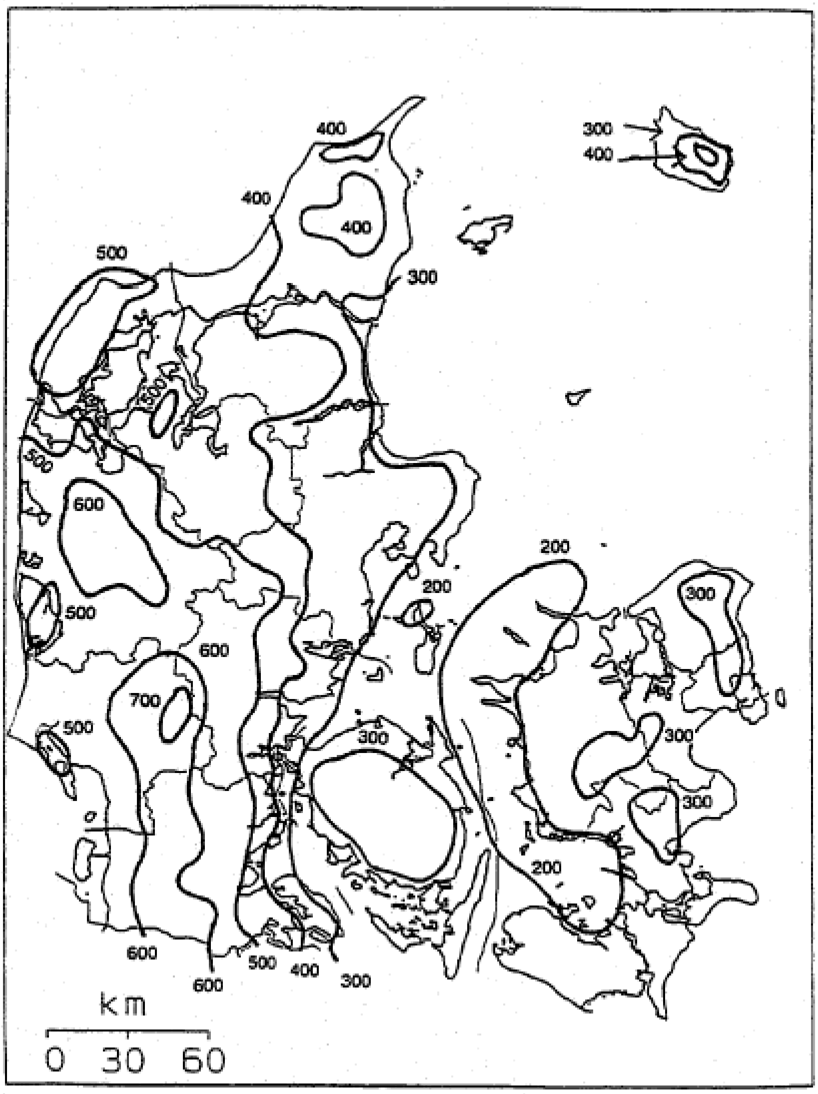

Figure 1
Net precipitation in Denmark (mm), mean 1961-90; /1/.

Deviations from the values shown here of up to approximately 40 mm per year are estimated to be common, but deviations of more than 60 mm will be rare.

2 Hydraulic conductivity and effective porosity

The hydraulic conductivity k is highly variable, and for this reason should always be determined at the specific site. Table 1 states typical values for horizontal hydraulic conductivity and effective porosity for various soil types /2, 5/.

In addition to this, certain known values for vertical permeability in clayey till are given /3,4/.

Table 1
Typical values for hydraulic conductivity (m/second) for various soil types /2,3,4/ and effective porosity /5/.

Material	Hydraulic conductivity, k (m/s)	Effective porosity, $e_{eff.}$
HORIZONTALLY:		
Clay soil (near the surface)		
Deep clay strata	10^{-8} - 10^{-6}	0.01-0.2
Silt	10^{-8} - 10^{-2}	0.01-0.2
Sand, fine	10^{-5} - 5×10^{-5}	0.01-0.3
Sand, medium grain	10^{-5} - 5×10^{-5}	0.1-0.3
Sand, coarse	5×10^{-5} - 10^{-4}	
Gravel	2×10^{-4} - 10^{-3}	0.15-0.3
Organic silt	10^{-3} - 10^{-2}	0.2-0.35
Sandstone	$\sim 10^{-10}$ -	0.1-0.35
Limestone	10^{-8} - 10^{-5}	
Rock, fissured and weathered	10^{-7} - 10^{-5}	0.1-0.4
	10^{-8} - 10^{-4}	0.01-0.24
VERTICALLY:		
Clay till		
1.0-1.5 m below ground level	1.3×10^{-5}	0.01-0.2
Clayey till		
2.0-2.5 m below ground level	4.2×10^{-6}	0.01-0.2
Clayey till		
4.0-4.5 m below ground level	2.5×10^{-7}	0.01-0.2

3 Hydraulic gradient

The hydraulic gradient, i, is not a standard parameter. The hydraulic gradient must be determined locally on the basis of water level measurements in investigation wells. Alternatively, the hydraulic gradient can be determined on the basis of local maps of the potentiometric surface.

4 Dispersivities

For risk assessments of groundwater, the longitudinal dispersivity α_L, cf. Section 5.4 and Appendix 5.6 of these guidelines, is used for calculating mixing thickness in the saturated zone (steps 2 and 3 of step-by-step risk assessment).

Figure 2 shows known values for the longitudinal dispersion as a function of distance. The size of the symbols indicate the reliability of the tests /2,6/.

Calculations of the mixing thickness d_m show that mixing thickness increases with dispersivity. The greater the longitudinal dispersivity, the greater the mixing thickness. For this reason, low values must be selected for α_L to ensure conservation calculations. Figure 2 shows the longitudinal dispersivity for a given distance on the solid curve.

5 Retardation factors

For sorbing substances, soil-water velocity V_p can in some formulae be replaced by the spreading velocity of the substance front V_s. The correlation between these factors can be described as (cf. Appendix 5.6):

$V_S = V_P/R$, where R is the retardation factor.

The retardation factor R is not a standard parameter, where a value applies to a larger geographic area.

The retardation factor depends on the substances involved, and on the bulk density of soil ρ_b, actual soil contents of organic substances f_{oc}, and on the octanol-water distribution coefficient K_{ow}. The content of organic substances f_{oc} for various soil types is found in Appendix 5.3, Table 1, and log K_{ow} values for various substances can be found in Appendix 5.5, Tables 1-5.

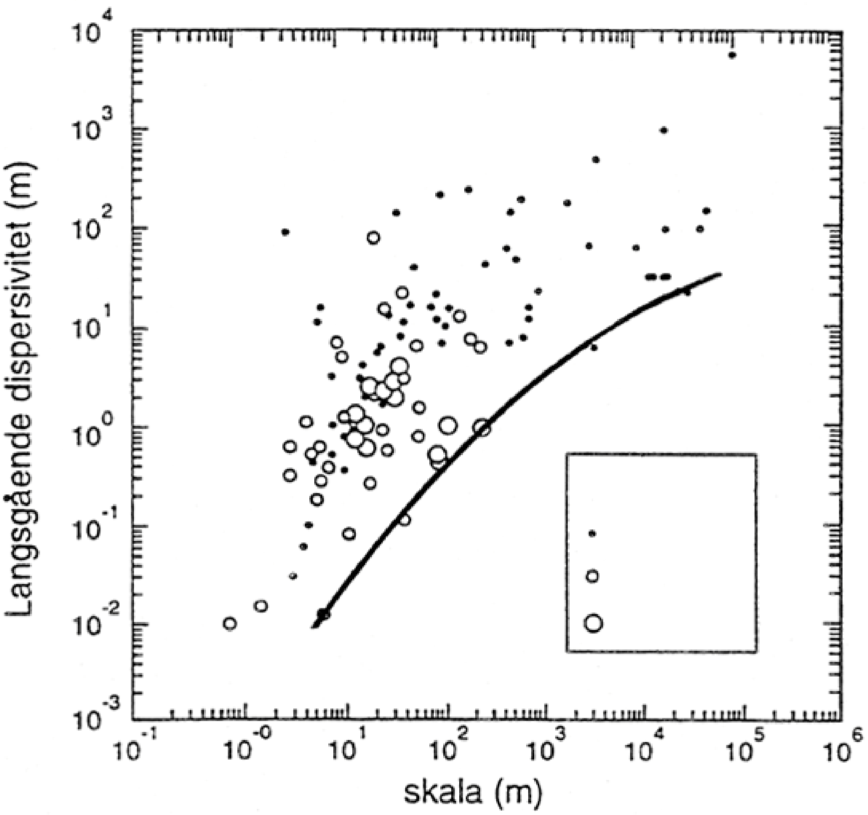

Figure 2
Longitudinal dispersivity as a function of distance /2,6/. The sizes of the symbols indicate the reliability of the tests.
For calculations of mixing thickness in saturated zones, α_L-values from the solid curve are used.

Given the assumptions that log $K_{ow} < 5$ and $f_{oc} > 0.1\%$, the distribution coefficient K_d can be calculated by means of Abdul's formula /1/:

$\log K_d = 1.04 \cdot \log K_{ow} + \log f_{oc} - 0.84$

The retardation factor can then be calculated by means of the formula:

$R = 1 + \rho_b/e_w \cdot k_d$

where ρ_b is the soil density $[ML^{-3}]$,
e_w is soil porosity when saturated with water [unitless],
and K_d is the distribution coefficient.

Examples of calculations of retardation factors are found in Appendix 5.7.

6 1^{st} Order degradation constants

As was outlined in Appendix 5.6, the relative substance concentration C on the basis of a 1^{st} order degradation can be calculated as:

$C_3 = C_2 \cdot \exp(-k_1 \cdot t)$

where t is the time period, during which degradation occurs [T],
C_3 is the resultant contamination concentration of the most contaminated zone of the groundwater aquifer after having taken degradation into account $[ML^{-3}]$,
C_2 is substance concentration before degradation$[ML^{-3}]$,
k_1 is the relevant 1^{st} Order degradation constant $[T^{-1}]$.

The degradation constants are substance specific, and moreover highly dependant on geological and hydrogeological conditions. For example, the degradation constants are often highly dependant on redox conditions. For many contaminants, degradation occurs fastest under aerobic conditions, other contaminants are exclusively degraded under anaerobic conditions, and some contaminants exclusively degrade under methanogenic conditions.

As yet, only very few examples of degradation constants determined in field conditions are available.

The degradation constants established so far vary greatly from one another. For this reason, it would be most favourable to determine the degradation constant at each site. Alternatively, conservative degradation constants must be used in calculations.

If calculations are made for degradation, it is important to ensure that there is potential for degradation throughout the entire period for the entire geographical area used in the calculations. For instance, in cases of aerobic degradation it must be ensured that oxygen is present throughout the entire period and the entire geographical degradation area. This is ensured by means of monitoring.

As part of a technology project, the Environmental Protection Agency has compiled 1^{st} order degradation constants which are deemed representative of Danish conditions /7/. Table 2 shows a compilation of these degradation constants.

Table 2
1st order degradation constants /5/; compiled after Kjærgaard et al /7/.

Contaminant	1st order degradation constant (day^{-1})		Comment
	Aerobic	Anaerobic	
BTEXs			
Benzene	0.01-0.2	0.001-0.003	Unlikely to be degradable in denitrified conditions
Toluene	0.05-0.2	0.01-0.1	
Ethylbenzene	0.01-0.1	0.002-0.03	Educated guess at aerobic degradation due to insufficient data
o-xylene	0.02-0.1	0.002-0.02	
m/p-xylene	0.001-0.02	0.002-0.03	
Chlorinated solvents			
1,2-dichloroethane	0	0.001-0.007	
1,2-dichloroethene	0	0.001-0.009	
cis-1,2-dichloroethene	0	0.0001-0.002	
Dichloromethane	0	0.0001-0.06	
Tetrachloroethylene	0	0.0005-0.004	
1,1,1-trichloroethane	0.005-0.006	0.0005-0.005	
Trichloroethylene	0	0.0001-0.008	
Trichloromethane	0	0.006-0.1	
Chloroethylene (Vinylchloride)	0.01*	0.0004-0.002	*Conservative estimate based on a single investigation
Other substances			
Phenol	0.07-0.4	0.001*	*Conservative estimate based on a single investigation

References

/1/ Mikkelsen, H. 1993. *Nettonedbør. Udkast*. ('Net Precipitation. Draft') Statens Planteavlsforsøg.

/2/ *Kemiske stoffers opførsel i soil og grundwater* ('Chemical Substance Behaviour in Soil and Groundwater') Projekt om soil og grundwater ('Project on Soil and Groundwater'), No. 20. The Environmental Protection Agency, 1996.

/3/ Jørgensen, P. R. (Danish Geotechnical Institute) and Spliid, N. H. (National Environmental Research Institute): *Migration and Biodegradation of Pesticides in fractured Clayed Till.*

/4/ *Grundvandsstrømning og udvaskning af forurening i moræneler. Geoteknisk Institut informerer.* ('Groundwater Percolation and Contamination Wash-out in Clayey till. Information from the Danish Geotechnical Institute') GI Info 5.8, 1993.

/5/ Wiedemeier, T.H. et al. 1996. *Technical protocol for evaulating natural attenuation of chlorinated solvents in groundwater.* Draft – Revision 1. Air Force Centre for Environmental Excellence, Technology Transfer Division, Brooks Air Force Base, San Antonio, Texas.

/6/ Gethar, L. W., Welty C. and Rehfeldt K. R. 1992: *A critical review of data on field-scale dispersion in aquifers*. Water Resources Research, 28, 1955-1974.

/7/ Kjærgaard, M., Ringsted, J.P., Albrechtsen, H.J. og Bjerg, P.L. *Naturlig nedbrydning af miljøfremmede stoffer i soil og grundwater* ('Natural Degradation of Alien Substances in Soil and Groundwater') the Danish Geotechnical Institute in collaboration with the Technical University of Denmark. A technology development project for the Environmental Protection Agency, 1998.

Appendix 5.9

Example of calculation of soil-water concentration based on soil concentration

The following provides an example of the use of the principle of fugacity to calculate the soil-water concentration of a contaminant on the basis of known soil contamination; i.e. a calculation of the soil-water phase transition.

The example given concerns benzene spill in clayey till. The values used in these calculations are the same values used in one of the examples of a risk assessment (Appendix 5.8 – calculation example 1).

1 Background

Calculations are carried out by means of the formulae given in Appendix 5.3 - section 3. The calculations are analogous to calculations in Appendix 5.4 - section 2, where calculations are carried out to determine contaminant contents in soil gas on the basis of know contaminant content in soil (the soil gas phase transition).

The following table shows values used in the calculation.

2 Calculations

The total volume of soil can be seen as the sum of the soil-phase volumes.

Equation 1

$$V_L + V_V + V_J = 1$$

where: V_L = relative volumetric proportion of air in soil (in this case 0.10),
V_V = relative volumetric proportion of water in soil (in this case 0.30),
and V = relative volumetric proportion of soil particles in soil (0.60).

Table 1
Parameters used in calculations

Relative volumetric proportion of air, V_L	0.10^*
Relative volumetric proportion of water, V_V	0.30^*
Relative volumetric proportion of soil, V_J	0.60^*
Soil temperature, T	281 K = 8° C
Soil particle density, d	2.7 kg/l*
Soil benzene concentration, C_T	1.0 mg/kg
Soil density, ρ	1.8 kg/l*
Soil content of organic matter, f_{oc}	0.001^*
Benzene partial pressure, p	12,700 N/m$^{2¤}$
Benzene molecular weight, m	78.1 g/mol$^¤$
Gas constant, R	8,314 J/mol·K
Benzene solubility, S	1,760,000 mg/m$^{3¤}$
Benzene octanol-water proportion, K_{ow}	$10^{2.1}$ l/kg$^¤$

*: cf. Appendix 5.3 - Table 1
¤: cf. Appendix 5.5 - Table 1

The maximum benzene content in one cubic metre (1 m^3) of soil distributed on the three phases of the soil can be calculated as follows:

In the air phase of soil (soil gas)

Equation 2

$$M_{L,\,max} = V_L \cdot C_{L,\,max} = 0.10 \cdot 425{,}000\ mg/m^3 = 42{,}500\ mg/m^3$$

where: $M_{L,max}$ = maximum benzene content in soil gas (mg/m^3 soil volume)
$C_{L,max}$ = saturated vapour concentration of contaminant (mg/m^3 soil gas).

By means of the law of ideal gases, $C_{L,max}$ is calculated on the basis of the partial pressure of benzene:

Equation 3

$$C_{L,max} = \frac{p \cdot m \cdot 10^3}{R \cdot T} = \frac{12700\ N/m^2 \cdot 78.1\ g/mol \cdot 10^3}{8.314\ J/mol/K \cdot 281\ K} = 425{,}000\ mg/m^3$$

where: p = benzene partial pressure (12,700 N/m^2)
m = benzene molecular weight (78.1 g/mol)
R = gas constant (8.314 J/mol · K)
T = temperature (281 K = 8° C).

In the water phase of soil (soil water)

Equation 4

$$M_{V,max} = V_V \cdot S = 0.30 \cdot 1{,}760{,}000\ mg/m^3 = 528{,}000\ mg/m^3$$

where:
$M_{V,max}$ = maximum benzene content in soil water (mg/m^3 soil volume)

S = benzene solubility in water (1,760,000 mg/m^3 soil water).

In the partial phase of soil

Equation 5

$$M_{J,max} = V_J \cdot d \cdot K_{oc} \cdot f_{oc} \cdot S \text{ (mg/m}^3\text{)}$$
$$= 0.60 \cdot 2.7 \text{ kg/l} \cdot 10^{1.344} \text{ l/kg} \cdot 0.001 \cdot 1{,}760{,}000 \text{ mg/m}^3$$
$$= 63{,}000 \text{ mg/m}^3$$

where:

$M_{J,max}$ = maximum amount of benzene adsorbed to the organic fraction of soil particles (mg/m^3 soil volume)
d = soil particle density (2.7 kg/l)
K_{oc} = benzene distribution between organic carbon and water (1/kg)
f_{oc} = soil content of organic carbon (0.001).

The distribution of benzene between organic carbon and water (K_{oc}) can be estimated on the basis of the octanol/water proportion K_{ow}, by means of Abdul's formula /1/:

Equation 6

$$\log K_{oc} = 1.04 \cdot \log K_{ow} - 0.84 = 1.04 \cdot 2.1 - 0.84 = 1.344$$

The maximum soil capacity for benzene (just before NAPL occurs), will then be:

Equation 7

$$M_{L,max} + M_{V,max} + M_{J,max}$$

$$= 42{,}500 \text{ mg/m}^3 + 528{,}000 \text{ mg/m}^3 + 63{,}000 \text{ mg/m}^3$$
$$= 633{,}500 \text{ mg/m}^3$$

Based on the assumption (cf. the concept of fugacity in Appendix 5.3) that the relative distribution of the three phases in soil is independent of on total concentration in soil, the distribution of benzene on the three soil phases can be calculated.

The following applies to the water phase of soil:

Equation 8

$$f_V = \frac{M_{V,max}}{M_{L,max} + M_{V,max} + M_{J,max}} = \frac{M_V}{M_L + M_V + M_J} = \frac{521{,}000}{633{,}500} = 0.833$$

where f_V = relative benzene proportion in soil water in relation to total soil content (calculated per m^3 soil).

M_L, M_V, M_J = actual amount of benzene in each of the three phases (mg/m^3 soil).

With a total concentration in soil C_T (in this example 1.0 mg benzene/kg soil volume) the amount of benzene in soil water M_V can then be determined as:

Equation 9

$$M_V = f_V \cdot C_T \cdot \rho = 0.833 \cdot 1.0 \text{ mg/kg} \cdot 1.8 \text{ kg/l}$$
$$= 1.5 \text{ mg/l soil volume}$$

where: C_T = benzene concentration in soil (1 mg/kg)
ρ = soil density (1.8 kg/l)

The benzene concentration in the soil water, C_V, is then calculated on the basis of the benzene concentration in soil, C_T.

$$C_V = \frac{M_V}{V_V} = \frac{1.5 \text{ mg/l soil volume}}{0.3} = 5.0 \text{ mg/l porewater}$$

References

/1/ *Kemiske stoffers opførsel i soil og grundwater* (‘Chemical Substance Behaviour in Soil and Groundwater’) *Projekt om soil og grundwater* (‘Project on Soil and Groundwater’), No. 20.
The Environmental Protection Agency, 1996.

Appendix 6.1

Calculated zero values for selected substances

Zero values are defined as calculated soil concentrations which correspond to soil-water concentrations may not exceed groundwater-quality criteria.

The calculations must be carried out in accordance with the principle of fugacity, as stated in Appendices 5.3 and 5.9.

Table 1 features calculated zero values for loam, sandy loam, clay, and sand (with soil parameters as stated in Appendix 5.3 – Table 1).

The zero values calculated are significantly below the soil quality criteria.

Table 1.
Calculated zero values for selected substances.

Soil parameters	Loam	Sandy loam	Clay	Sand
Volumetric proportion (%)				
- soil	60	55	60	55
- air	10	10	10	30
- water	30	35	30	15
- organic matter	1.0	2.0	0.1	0.2
Particle density (kg/l)	2.65	2.6	2.7	2.65
Soil density (kg/l)	1.7	1.6	1.8	1.7
Contaminant	Zero values (µg/kg)			
Benzene	0.4	1.2	1.9	0.2
o-xylene	21	109	5.0	13
m-xylene	59	96	7.7	11
p-xylene	61	114	9.1	13
Naphthalene	253	477	33	50
Dichloromethane	1.7	2.2	1.4	0.9
Trichloromethane	2.3	3.6	1.3	0.1
Tetrachloromethane	0.8	1.4	0.3	0.4
1,1-dichloroethane	1.1	1.7	0.8	0.6
1,2-dichloroethane	4.6	6.4	3.6	2.2
1,1,1-trichloroethane	1.1	1.8	0.4	0.4
Chloroethylene	0.02	0.03	0.02	0.04
1,1-dichloroethylene	0.5	0.7	0.3	0.3
Trichloroethylene	2.1	3.6	0.7	0.7
Tetrachloroethylene	2.3	4.2	0.5	0.7

Eco-toxicological risk assessment

An eco-toxicological risk assessment of contaminated sites is carried out in steps, so that the initial assessment is based on existing data or a comparatively simple screening. If a risk is identified, the investigations can then be extended in one or more steps. However, special information on non-sensitive area use may render further assessment obsolete.

Stage 1
The results from chemical analyses are compared with eco-toxicological criteria of soil quality /1,2/. The contamination is unproblematic as regards the soil environment if the criteria for all substances are met, and high concentrations of substances for which no criteria exist are not identified at the site in question.

Stage 2
If a correlation of chemical analyses and eco-toxicological soil-quality criteria pose a risk to the terrestrial environment (plants, soil-dwelling animal life, or micro-organisms), the assessment is supplemented by biological investigations of the uppermost soil strata at the site. If the contaminated area is situated in an urban environment, the investigations at stage 2 may be omitted. If the contamination consists of few substances, or of mixtures of related substances in concentrations which must be deemed high in terms of environmental impact, it is sufficient to examine the effects on the most sensitive group of organisms, if this group is identifiable.

The following test methods are recommended:

Micro-organisms

- Soil respiration (a general parameter for carbon decomposition in soil: OECD draft or ISO 14240).

- Inhibition of ammonium-oxidising micro-organisms (a specific parameter for the nitrogen cycle; OECD draft or ISO 14238).

Plants
The OECD standard test for plants (OECD No. 308 or ISO/CD 11269).

Soil-dwelling animal life:

- OECD earthworm test (acute toxicity, OECD 207 or ISO 11268-1 (DS/ISO)).

- Collembolan test (reproduction test; ISO/CD 11267).

If other internationally recognised methods or method suggestions are used, the reasons for this should be stated in the reports.

Stage 3
If one or more tests at stage 2 show effects, it is recommended to carry out a charting of the species composition of plants and/or soil-dwelling animal life at the site with a view to assessing effects on the site.

Stage 4
If, on the basis of the laboratory tests at stage 2 and/or the field investigations at stage 3, the risk assessment displays significant impact on flora, fauna, or micro-organisms, an assessment of the acceptable risk can be carried out. With consideration being given to future area use, proposals can then be prepared for special measures to protect flora, fauna, and micro-organisms, if such protection has not already been achieved by means of measures to protect human health, groundwater, or the surface recipients.

References

/1/ *Økotoksikologiske jordkvalitetskriterier* ('Eco-toxicological soil-quality criteria'). Working report from the Environmental Protection Agency. No. 82, 1997.

/2/ *Økotoksikologiske jordkvalitetskriterier* ('Eco-toxicological soil-quality criteria'). Project on soil and groundwater from the Environmental Protection Agency. Report No. 13, 1995.

Appendix 7.1

Reporting

Figures and Tables

Figures are included to facilitate comprehension and to provide overviews. Remember a key of symbols and to state scale and compass on all maps.

Maps are more easily readable if round scales are used; 1:25,000, 1:10,000, 1:500, etc.

In some cases, the extent of contamination can be presented graphically by means of bar charts or pie charts. However, this method of presentation has a weakness in that it does not provide opportunities for outlining the three-dimensional extent of contamination.

By outlining a number of simple geological sections on which the results from analyses are stated, it is possible to present an image of the contamination in three dimensions. These sections may take up so much space that they have to be placed in an annex.

Below is a list of the figures which may be included in a preliminary contamination investigation at a site:

1.1 Overview: Location of the site.
2.1 Site plan: Position of buildings and plants during the period of operation.
2.2 Site plan: Present landuse.
2.3 Overview: Location of the site in relation to water-supply wells and surface recipients.
4.1 Site plan: Location of borings, soil gas measuring points, etc.
5.1 Site plan: Potentiometric surface maps for groundwater aquifers near the surface at the site.
5.2 Site plan: Potentiometric surface maps for groundwater aquifers at greater depths in the area.
6.1 Site plan: Extent of contamination – soil.
6.2 Cross-section: Extent of soil contamination.
6.3 Site plan: Extent of contamination - groundwater near the surface.

A number of the above site plans may take up so much space that they have to be placed in an annex.

If more than a few numbers are to be presented in a text, this is best done in a table.

Text in tables and figures should be short, yet sufficiently descriptive to render the table/figure immediately comprehensible.

Appendix and annex

Below are examples of the types of appendices the report can include:

- Drilling work and collection of soil samples
- Collection of water samples
- Soil gas measurements
- Measurements with photo-ionisation detector
- Analytical methods and detection limits

Stated below are the types of annexes which may be included in a contamination investigation:

- Site plan with location of borings and measuring points
- Lithologic logs, including key to symbols
- Levelling datums and water-level measurements
- Geological cross-sections
- Documentation of water sample collection
- Analytical reports

The number and size of appendices and annexes depends on each task, and the above list of appendices and annexes can be supplemented as needed.

Tender materials

The tender material must include a detailed description of the remedial project. Below, an outline for the tender materials is provided:

- Letter of invitation to tender
- Contractor summary
- Normal conditions (NC)
- Special conditions (SC)
- Special work descriptions (SWD)
- Tender and settlement (TS)
- Tender list (TL)
- Drawings
- Annex

The letter of invitation to tender should include a list of the tender materials as well as deadlines for submissions.

The summary forms part of the tender material and may conveniently form the background of Special Work Descriptions (SWD). The summary is intended to be for information only, and as such cannot replace documents in the tender material. The contractor summary must include:

- Type of remedial measures.
- Contamination composition and extent (the volume of soil or groundwater to be removed/treated is stated).
- Exceptional measures in connection with the remediation project (lowering the groundwater table, water treatment, bunging, etc.)
- Type and extent of technical installations to be removed in order to carry out the project (pipes, cables, tanks, etc.).
- Restoration form.

Normal conditions describe general conditions for work and supply. In Denmark, AB92 is generally used /35/.

Special conditions covers the variations and amendments which are applicable in addition to normal conditions, AB92.

Special conditions should include information on:

- Briefing
  A general presentation on the case and the developer is provided. Sub-sections could include:
    - Project organisation, i.e. developer, building administration, contractors, etc.
    - Site location (address and title number)

  - Project description (brief)
  - Basis for the contract (scope of project material)
- Basis for agreement (normal conditions, developer invitation to tender, building contractor tender).
- Collateral and insurance (standard form for collateral is attached).
- Execution of contract (workplan and subcontracts, project review, relations with authorities, changes, unclear points, obstacles or similar conditions, building-contractor management, site meetings, any collaboration with other contractors).
- Developer liabilities to pay.
- Deadlines and delays (including fines).
- Completion of work.

Works descriptions are instructions for the building contractors and include all work and supplies in connection with the execution of the contract. The work descriptions are often designated ‘Special Work Descriptions’. The work descriptions should include information on:

- Background (description of the relevant contamination problem with a view to determining purpose, working environment, description of remedial project, etc.).
- Workplace organisation and operation (work area; position of site huts, materials, access conditions, etc.; workplace organisation and operation; geo-technical information; existing pipelines etc.; responsibilities, rights, and obligations; time schedule, safety precautions; safety descriptions; assessment of relevant hazardous substances, etc.).
- Clearing (general, reusable objects, grass, soil, plants, etc.).
- The execution of work (description of the series of requirement specifications made for individual projects).
- Restoration (description of requirements of the appearance of the site upon work completion, as regards paving, installation, plants, etc.).
- Possible preparation of quality-assurance manual.

Conditions and guidelines for filling out the tender list, and comments on individual items on the tender list. With most remedial projects, the basis for tender and settlement is small enough to be most appropriately included in the tender list.

Tender lists stating individual activities, any reservations, person or persons with authority to negotiate, any alternative tenders, and any sub-contractors.

Drawings relevant to the tender material should be enclosed.

Annexes of relevant information are enclosed (investigation reports, lithologic logs, time schedules stating deadlines).

Appendix 9.1

Prevention techniques and financial examples

Table 1
Remediation techniques.

Method	Contamination type			Soil type	Documentation e)	Other conditions
	Org./ Inorg. a)	Volatility b)	Degrada-bility c)	Permeability d)		
Soil contamination						
Excavation with off-site treatment	+/(-)	All	+	All	++	f)
Excavation with landfilling	+/+	All	All	All	++	g)
Excavation and on-site treatment	+/-	+	(+)h)	(+)h)	+	i)
Soil vapour extraction	+/-	++	-	+	++	
Bioventing	+/-	+	++j)	+	+k)	
Forced leaching	+/+l)	-	-	++	+	m)
Immobilisation	+/+	(+)n)	All o)	All	+	
Steam stripping	+/-	+	All	+	(+)p)	q)
Groundwater contamination						
Remedial pumping, draining	+/+l)	All	All	+	++r)	s)
Bio-slurping (including suction probes)	+/+t)	All	All	+	+	
In-situ remedial methods for groundwater contamination						
Air sparging	+/-	+u)	-v)	+	+	x)
Adding oxidising agents (ORC)	+/-	All	+	+	(+)y)	
Vertical cut-off barriers	+/+	(+)n)	All	All	+	
Reactive permeable barriers	+/+	All	+z)	+	(+)y)	æ)
Natural attenuation	+/-	All	+ø)	All	(+)y)	å)

a) +/- = organic contamination
-/+ = inorganic contamination
+/+ = both types
b) ++ = very volatile
+ = volatile
- = non-volatile
c) ++ = very degradable
+ = degradable
- = non-degradable
d) ++ = very permeable
+ = permeable
- = very low permeable
e) ++ = very well documented
+ = tested in Denmark
- = effect not documented
f) Contamination position is vital
g) High environmental impact
h) Depending on method of cleansing
i) Makes great demands on surroundings etc.
j) Aerobically easily degradable substances
k) A number of plants in active use
l) Demands substance water-solubility
m) May cause problems of plant clogging
n) Depending on method, but usually chosen for high-boiling contamination
o) This method is usually selected for contamination which is difficult to degrade
p) Not used in DK
q) Requires level ground and that no rocks are found above a soil depth of approximately 0.3 m. Energy-intensive
r) Good for hydraulic contamination control, but may be difficult to obtain with low acceptance levels
s) Attention must be given to problems in connection with substances with a density greater than that of water
t) Particularly useful for NAPL oil contamination
u) Stripping of contamination must be possible
v) Contamination must be degradable with bio-sparging
x) Contamination must be removed from the unsaturated zone, possibly by means of soil vapour extraction
y) Effects proven in the USA
z) Not necessarily aerobically degradable contamination
æ) Used in combination with ‘Funnel & Gate'
ø) Contamination must be proven to degrade
å) This method demands extensive monitoring

Table 2
Remediation techniques and financial examples.

METHOD	COSTS, excluding VAT
Soil contamination	
Excavation	The cost of excavation and transport (within a radius of 150 km excluding transport by water) will amount to approximately DKK 90-200/tonne for a normal excavation, depending of the amount of soil, contamination position, transportation distance, and geographical position. The cost of filling the excavated area back up is approximately DKK 60-100/tonne depending on amount, type, and requirements for compactness. Additional costs include restoration, environmental inspection, documentation, decontamination, and any monitoring. Restoration costs vary greatly, depending on physical conditions and the type and strength of contamination. Experience shows that with oil/petrol contamination; the cost of contamination excavation will amount to DKK 400-1,000/tonne (including excavation, decontamination, and back-filling of new or treated soil) in cases where more than 1,000 tonnes of soil are excavated. For smaller contamination projects involving up to 1,000 tonnes of soil, costs will typically fall between DKK 400-2,500/tonne (normally DKK600-800 /tonne).
Soil treatment	The cost of soil treatment at central treatment plants is approximately DKK 160-600/tonne for biological soil treatment, depending on contamination type and amount (lighter types of contamination are cheaper). The cost of incineration is typically DKK 600/tonne at power plants and up to DKK 1,000/tonne at treatment plants (heavy organic contamination, e.g. tar), whereas thermal treatment for very contaminated soil can cost up to DKK 3,500 – 4,000/tonne (excluding heavy metals).
Landfilling	The cost of disposal at landfill sites varies greatly. However, the cost normally included a waste tax, which currently amounts to DKK 335/tonne (1997). Soil which is contaminated by heavy metals may in some cases (weak contamination – up to contamination level 3) be deposited at special landfills. The cost is approximately DKK 130 – 150/tonne. Prices of DKK 450-800 (including waste tax) per tonne must be expected for disposing of soil contaminated by heavy metals at landfill sites, depending on geographic position and level of contamination (up to contamination level 3). Soil which is heavily contaminated by heavy metals (corresponding to level 4) can be deposited at landfills or sent to KOMMUNEKEMI at a cost of approximately DKK 1,000/tonne.
On-site treatment of excavated soil	The cost of on-site treatment varies according to the method and the composition of the contamination. Prices of DKK 50-150/tonne have been observed for landfarming, whereas prices of up to DKK 1,000/tonne have been observed for mobile thermic on-site plants (heavier contamination).

METHOD	COSTS, excluding VAT
Soil vapour extraction	This method is relatively cheap. A standard plant comprising 5 borings (63 mm diameter to a depth of 4 m), piping, ventilation with discharge, noise reduction, and electric wiring normally costs less than DKK 150,000. Cost of operation and monitoring (electricity, inspection, and final documentation) must be added to this. For the plant outlined above, these costs will amount to approximately DKK 50,000 a year. If the exhaust air is to be treated, approximately DKK 30,000 must be added to cover set-up (carbon filters for air flows of up to 600 m^3/h). Operation costs will also be increased due to carbon-filter replacements. For example, with an airflow of 500 m^3/t, the lifespan of a carbon filter with 260 kg of carbon will be approximately 60 days for 50 mg/m^3 benzene, 110 days for the same amount of toluene, and 120 days for the same amount of trichloroethylene. This corresponds to annual costs of carbon-filter replacement of approximately DKK 175,000; 95,000; and 90,000, respectively. Attention should be given to the fact that concentrations drop significantly at the beginning, causing similar reductions in the cost of treatment of discharged air. The plant outlined above would typically cover a contaminated area of 200-500 m^2; depending mainly of the type of soil.
Bio-ventilation	In terms of set-up, this method is inexpensive. However, the tasks of monitoring and inspection are comprehensive, leading to high operation costs. The prices are comparable to soil vapour extraction.
Forced leaching	A plant including a pumping well (to a maximum depth of 15 m), distribution well, leach field (20 m), placement of pumping plant with level control, and piping can be set up for less than DKK 100,000. This does not include decontamination and screening of pumped-up water (see Section 9.3.5). Operation costs must also be added to this; these costs mainly involve plant inspection and monitoring.
Immobilisation	Costs should be assessed in each case, depending on what is to be done. Normal surface sealing using asphalt or paving is a relatively cheap solution, as this can often be combined with a building project. Examples of costs of approximately DKK 700/tonne are observed for stabilising.
Steam stripping	This method has not been attractive in Denmark, primarily because of the great costs of this very energy-intensive method (examples from the USA indicate prices of 1,200 to 3,400/tonne). Moreover, this method requires that all objects larger than approximately 0.3 m be removed from the soil and that the terrain does not slope more than 1 %.

METHOD	COSTS, excluding VAT
Groundwater contamination	
Remedial pumping	The cost of construction depends on the technical layout. For example, the cost of a unit with a 10" well well to a depth of 20 m with a 160 mm screen diameter, complete with dry well, pump, raw-water station, electricity, and discharge will be approximately DKK 70,000 – 90,000. Any costs involving treatment units, operation, etc. must be added to this figure.
Drainage	With simple cases, drainage can be established at a cost of approximately DKK 300 per metre.
Suction probes	Suction probe equipment often comprises 20 suction probes to a maximum depth of 6 m and can be established for less than DKK 15,000. It is often advantageous to hire the vacuum facility due to limited time scales. Hiring a facility such as the one outlined above amounts to approximately 4,000 a week.
Bio-slurping	
On-site treatment of abstracted groundwater	
Discharge	Taxes impose significant expenses on discharge of abstracted groundwater to sewers. This tax varies among local authorities. The diversion tax typically amounts to DKK 10-15 per m^3 (1996).
Oil/water separators	A traditional gravimetric oil/water separator can be purchased for DKK 10,000 – 20,000 depending on size, whereas coalescence oil/water separators cost more than three times as much.
Coalescence separators	Coalescence separators can be purchased for DKK 50,000 – 100,000. Cleaning water by passing it through screens is a well-documented method. The choice of material depends on the actual contamination situation.
Filters	The cost of setting up a double carbon-filter plant for a water flow of up to approximately 5 m^3/h amounts to DKK 70,000-80,000. This does not include any pre-filter, nor any containers, etc. The operation costs depend on the relevant contamination situation (carbon-filter replacement). For example, at a water volume of 2 m^3/hour, the replacement frequency of two 450 kg carbon filters is approximately once a year for water concentrations of 1 mg of benzene/l; once every two years for 1 mg of toluene/l, and once every two years for 1 mg of trichloroethylene/l. This corresponds to annual costs of approximately DKK 70,000; 35,000; and 35,000 respectively, not including inspection and monitoring.
Air stripping	The cost of setting up an air stripping plant is relatively high, approximately DKK 100,000 – 150,000. Careful control and monitoring must be added to this figure. In addition to this, pre-filtering must normally be included in calculations, just as a subsequent treatment of the air is often required.

METHOD	COSTS, excluding VAT
Photochemical oxidation	Plant investments amount to approximately DKK 80,000 – 150,000 depending on design. Operation costs mainly comprise the oxidation agent (hydrogen peroxide), lamp replacement, electricity, and regular inspection.
In-situ remediation methods for groundwater contamination	
Air sparging	The cost of a standard plant, comprising a system of 10 remedial wells to depths of approximately 10 m, including dry well, piping, containers with noise reduction for the operation unit, and muffled air intake, including the cost of air treatment, amounts to approximately DKK 1 – 1.5 million. This does not include the cost of soil vapour extraction from the unsaturated zone. Moreover, relatively high operation and monitoring costs must be expected due to energy consumption and the need for inspection; for the plant outlined above this will typically amount to DKK 200,000 per year. Wells are usually positioned at intervals of approximately 10 m, so the plant outlined in the above would cover a contaminated area of approximately 1,000 m^2.
Adding oxidising agents (ORC)	The method is cheap and environmentally sound, and is likely to become widespread in Denmark.
Vertical cut-off barriers	It is possible to cut off groundwater contamination by establishing vertical barriers in the groundwater aquifer. This is done by means of various methods, such as sheet piling, deep soil mixing (DSM), slurry walls, and grouting.
Reactive permeable barriers	Reactive permeable walls are barriers which allow the groundwater to pass, but which degrade or remove contamination during passage. This method is at the experimental stage in Denmark, but is used in field work in the USA with degradation of chlorinated compounds by adding iron to the barriers.
Natural attenuation	This ‘method’ shows promise, as the resulting environmental impact is very low. Consequently, it may have a future, especially as regards clean-up of oil/petrol contamination.
Precautions against landfill gas	Preventing gas from spreading by means of interceptive methods can entail relatively high costs. For example, the initial investment to render a site safe by means of interceptive drainage of approximately 100 m to a depth of approximately 4 m will be approximately DKK 500,000, whereas the annual operation and monitoring costs amount to approximately DKK 15,000 – 20,000. With new building, the constructional precautions form a very small part of the total cost when taken into consideration during the initial stages of the project.

Excavation near existing buildings

With excavations of any kind, for example in connection with contamination excavations, care must be taken to avoid weakening the stability of existing buildings. Failure to do this may in extreme cases lead to construction failure. This issue is addressed in *Byggeloven* ('the Building Act') in Section 12, 1.

Experience shows that damage rarely occurs when the consultant/ developer/contractor responsible for the project has the geotechnical assistance which is required to deal with the problems. However, damage often occurs when financial considerations or pressures of time force a project to take place at the limit of what is safe, and when there is simple ignorance of actual conditions.

Typical examples of buildings which may damage existing constructions in connection with excavation are:

- free excavation next to existing buildings with no basement
- drainage projects next to existing buildings with foundations on sand
- underpinning existing foundations

The excavation project must be designated as belonging to foundation class 1 (relaxed), 2 (normal), or 3 (strict).

Temporary excavations in foundation class 1 (relaxed) can usually be carried out, provided that excavation occurs above the groundwater table in late-glacial or older sediments (not fissured, sticky clay), and the construction at risk has foundations in accordance with normal requirements (no strict foundation class). In such cases, excavation can be done within the areas shown in Figures 1 and 2.

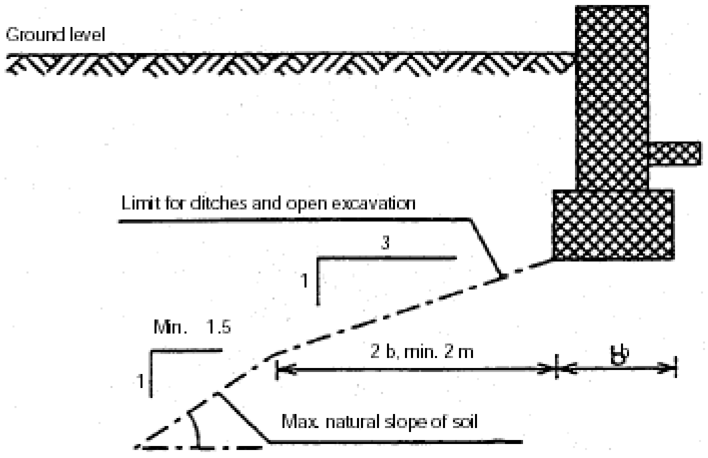

Figure 1
Excavation in clay above the groundwater table

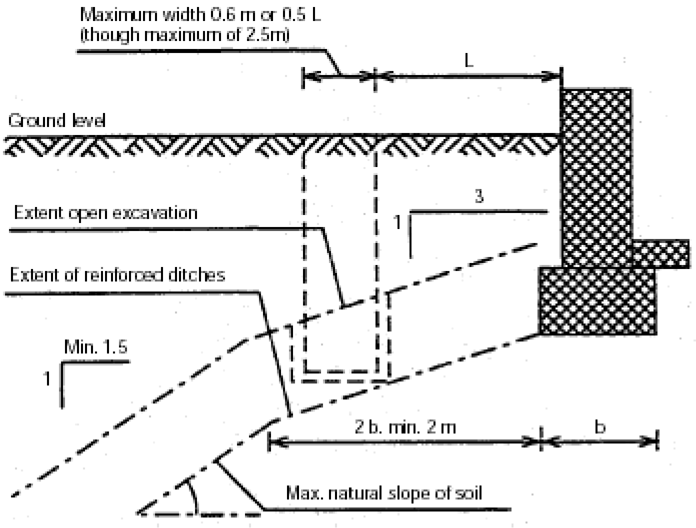

Figure 2
Excavation in silt, sand, and gravel above the groundwater table.

If these prerequisites are not met (verified by means of individual test holes along the construction), the necessary geotechnical tests must be carried out to determine soil and groundwater conditions. This may render it possible to carry out deviations from the excavation levels indicated, and to provide the guidelines necessary to carry out the task in foundation class 1 (normal). Calculations of foundation bearing capacity at the edge of excavations may e.g. be carried out in accordance with standard foundation specifications, DS415.

Excavation complications in the form of sticky, fissured clay; adjacent constructions with single foundations, or eccentric stress will automatically place excavation projects in foundation class 3 (strict).

All excavation in the vicinity of existing constructions must occur above the groundwater level. This means that high water tables must be drawn down before excavation is carried out.

Many excavation projects, for examples projects which take place in the immediate vicinity of houses without basements, require excavations that must go substantially further than the extent indicated in Figures 1 and 2. In such cases, excavations by section is common. The basic concept of this method is that existing foundations are only weakened along one (or several separate) very limited subsections at a time.

However, accidents frequently occur in connection with excavations by section, especially when work is conducted in sandy types of soil. For this reason, the rules outlined below should always be followed:

- The building contractor must be under obligation to carry out excavations in accordance with a plan which ensures that neighbouring foundations are not weakened.
- This plan must indicate the maximum lengths and depths of individual excavation sections, as well as which sections may be excavated simultaneously.
- Work must take place above the groundwater table. Any drawdown of the groundwater must only be done by means of draining from the excavation sections themselves in a minority of cases. Normal conditions require pumping or draining at a safe distance from the existing foundations to avoid loss of material below the foundation.
- When underpinning serves to facilitate basement excavation next to a building without a basement, planning should include proof that the new foundation is stable (vertically and horizontally), taking into account e.g. pressures of soil and water to the backside of the foundation.
- With foundations subjected to local weakening due to excavations, the extent of this weakening can be estimated as illustrated by Figure 3. Notice how weakening is significantly greater with sand in comparison with clay; a fact which is also clearly reflected by accident data.

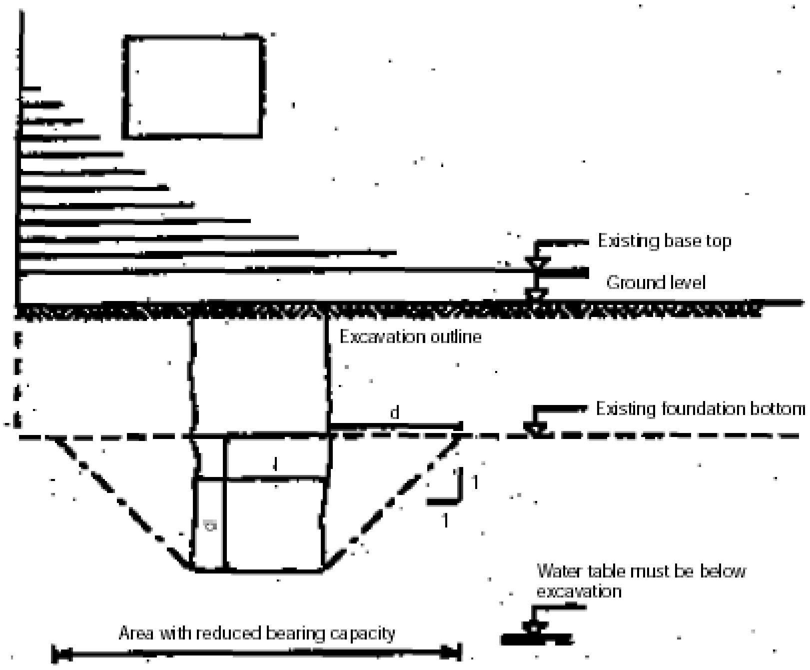

Figure 3
A distance of the length l + 2d must be expected to be reduced in bearing capacity when excavating existing foundations. The remaining bearing capacity ($\frac{Q}{b}$ kn/m) can be determined as follows (for linear foundations with central stress).

Along the excavated distance
l ≤ 1.2 m) is $\frac{Q}{b} = 0$

The bearing capacity is approximately halved along the two sides (d).

If the sand has no capillary tension (coarse and medium grain sand), the bearing capacity for the two sides (d) must be expected to be reduced to 0. Reinforcement wedged up against the sides of the excavation may improve this considerably. In this case, reinforcement must be dimensioned to correspond to the required vertical bearing capacity along the sides (d).