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Risk assessment framework for management of the factor 3 rule in the Council Decision 2003/33/EC on waste acceptance criteria

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Content

Foreword

This document is the final report on the project entitled: “Risk assessment framework for management of the factor 3 rule in the Council Decision 2003/33/EC on waste acceptance criteria” financed by the Nordic Council of Ministers, the PA-Landfill group. The project was initiated in October 2006 and finished in June 2008.

Certain waste types (e.g. air pollution residues from flue gas cleaning in MSWI plants) cannot comply with the EU leaching criteria for waste to be accepted at landfills. Council Decision 2003/33/EC provides a possibility to exceed these criteria by a factor of three under certain conditions. The project addresses the required evaluation of impacts to soil and water relating to this deviation from EU acceptance criteria for wastes to be landfilled i.e. the application of the “factor 3 rule”.

The main objective of the project has been to achieve a common approach/understanding in the Nordic countries relating to deviations from EU acceptance criteria for landfilling. An important part linked to the project was a Nordic workshop held on October 29 and 30, 2007. Nordic stakeholders and specially invited European experts were asked to comment the draft project report and highlight key factors to be considered in a risk assessment, to share experience from modelling of release and also to pinpoint current needs (e.g. critical waste streams not complying with the EU waste acceptance criteria for landfilling). The participants were also asked to identify special demands on landfill operations in case of deviation. The workshop conclusions are included in this report.

The project has been carried out by VTT from Finland in cooperation with its Danish partner DHI.

Abstract

The Council Decision 2003/33/EC on waste acceptance criteria related to the EU landfill directive 1999/31/EC establishes acceptance criteria for wastes (primarily limit values for the leaching of predominantly inorganic components) to be disposed of at landfills for inert waste and hazardous wastes and also for stable, non-reactive hazardous waste to be placed in non-hazardous waste landfills (and for non-hazardous waste to be placed in the same cells). Criteria for the other types of landfills and for contaminants not included in the EU regulations are to be developed on a national basis.

The Council Decision allows member states the possibility on a case-by-case basis, to allow up to three times higher limit values for specific wastes and specific components. However, the use of higher limit values requires a risk assessment to demonstrate that there will be no additional risk to the environment. The requirements for the landfill construction and leachate collection can also be reduced on the basis of an assessment of environmental risks. It has then to be shown that the landfill poses no potential risk to soil, groundwater or surface water. The same kind of impact assessment will be relevant also in cases where national waste acceptance criteria may be set for contaminants that are not included in the EU Council Decision.

The need for higher acceptance values, especially for leaching of salts from waste from waste incineration, has been raised in the Nordic countries. At present, there is no guidance on how to perform a risk assessment in this context. Also the possibilities to deviate from the politically defined and in some cases increased limit values require discussions and background information.

The aim of the study is to provide guidance on possible approaches to a risk assessment in cases where deviations from the landfill regulations are considered and to point out key issues to be taken into account. A proposal for an approach to be used in the impact assessment is presented. The focus is primarily on inorganic constituents. The results of this project may be used in the development of national guidelines on deviation from EU landfill criteria.

Definitions

scenario	description of a set of normal and exceptional conditions relevant to a particular disposal or utilisation situation for waste for the determination of the leaching behaviour within a specified time frame (EN 12920)
risk	combination of the probability of the occurrence of a hazard and the severity of that hazard
risk analysis	use of available information to identify hazard and to estimate the risk
risk assessment	process of risk analysis and risk characterisation
risk characterisation	evaluation and conclusion based on the hazard identification and the exposure and the effect assessment
hazard	inherently dangerous quality of a substance, procedure or event
groundwater	all water which is below the surface of the ground in the saturated zone and in direct contact with the ground or subsoil
surface water	inland waters, except groundwater; transitional waters and coastal waters, except in respect of chemical status for which it shall also include territorial waters
impact assessment	process of analysis of the influence (quality/quantity) of an acitivity on a target (e.g. air, soil, groundwater, surface water, humans, the ecosystem, animals, plants, etc.
release	emission of constituents from a waste which pass through the waste external surface of a waste mass in the considered scenario
WAC	waste acceptance criteria defined for acceptance of waste to certain landfill categories
GWT	ground water table
POC	point of compliance, the location where the impact is evaluated based on a defined acceptable risk level

(most of the definitions from ISO/TC190 “Soil quality”, EN 12920 and the Water Framework Directive 2000/60/EC)

1. Introduction

1.1 Background

The Council Decision 2003/33/EC on waste acceptance related to the EU Landfill Directive 1999/31/EC provides a possibility to increase the limit values for leaching of predominantly inorganic components with up to factor of three. The need for higher acceptance values, especially for leaching of salts from wastes from waste incineration, has been raised by some of the stakeholders in the Nordic countries.

Today, no guidance exists on how to approach the risk assessment required if or when the increased limit values are to be applied. Deviations from the general EU landfill regulation require knowledge of waste characteristics and landfill scenarios as well as impact assessment modelling. The number of experts in the Nordic countries with expertise both on waste characteristics and waste impact modelling is limited, and the possible deviation from the EU limit values, some of which have been defined politically and increased substantially compared to the risk assessment upon which they were based, require discussions and consideration of background information. There is therefore a need for exchange of knowledge and information between the Nordic countries. Furthermore, a common approach to the risk assessments will make the possible acceptance of deviations more consistent within Nordic countries.

1.2 Objective

The aim of the study has been to provide guidance on possible approaches to a risk assessment in cases where deviations from the landfill regulations are considered and to point out key issues to be taken into account. A proposal for an approach to be used in the impact assessment is presented. The focus is primarily on inorganic constituents. The results of this project may be used in the development of national guidelines on deviation from EU landfill criteria.

The report also seeks to address the need expressed by Nordic stakeholders for deviations for specific wastes. The target group for this report includes all stakeholders involved such as authorities, regulators, waste producers, consults and also testing laboratories.

This report does not consider the influence of the waste on liners, leachate treatment and aftercare. Neither does it address aspects related to occupational health at waste facilities.

The study focuses on the possibilities for deviation from the waste acceptance criteria for waste streams generated in significant amounts. The baseline is a monolandfill for predominantly inorganic waste because it is at present not possible to model or estimate the leaching behaviour of waste in landfills for nonhazardous waste containing significant amounts of organic, biodegradable waste. This is due to the complex nature of the biological processes occurring in the landfill. It is suggested that a methodology for impact assessment similar to the one described here can be applied to inorganic wastes utilised for construction engineering purposes.

Parallel to and in support of the project, a twoday workshop on the factor 3-rule was held on October 29-30, 2007 at VTT in Espoo, Finland. Experts from all Nordic countries were invited to the workshop. The workshop programme and participant list are presented in Appendix 1. The main results of the workshop have been collated in Appendix 2. The aim of the workshop was to discuss the possibilities for deviations, to address needs in the Nordic countries and to obtain comments from landfill operators on future demands in landfill operations in case of deviations of EU landfill criteria. The intention was also to encourage further Nordic work and collaboration in this area.

1.3 Project organisation

This project has been carried out by VTT and DHI. Co-operation and exchange of information took place with Ebba Åkerlund from SGI in Sweden who managed a related national project in 2007. The project group also consulted Dr. Jan Gronow (chairman of the EU sub-TAC-group developing the EU limit values) for discussions on the interpretation of the EU Council Decision and possibilities for derogations from the legislation.

VTT was responsible for the overall management and dealt with Nordic key issues to be considered in the risk assessment and aspects linked to evaluation of acceptable risk. The modelling work was carried out by DHI with input from VTT (particularly on selection of assumptions and points of compliance). SGI took part in the discussions of test results and issues to be considered in the modelling work. DHI participated in the original modelling work for setting of the European criteria for acceptance of waste at landfills. VTT and DHI worked out recommendations for possible approaches for deviations from the waste acceptance criteria. The final language check of the report has been performed by DHI.

The project group has included the following persons:

Partner	Institute
Margareta Wahlström	VTT, Finland (project manager)
Jutta Laine-Ylijoki	VTT, Finland
Ole Hjelmar	DHI, Denmark
Erik Aagaard Hansen	DHI, Denmark

2. Legislative boundaries

2.1 The EU Landfill Directive: Principles for acceptance of waste to landfill

The general principles upon which the criteria for acceptance of waste at landfills should be based are described in Annex II to Directive 1999/31/EC of 26 April 1999 on the landfill of waste. In Annex II (article 2) it is stated that:

"Criteria for acceptance [of waste] at a specific class of landfill must be derived from considerations pertaining to:

- protection of the surrounding environment (in particular groundwater and surface water),
- protection of the environmental protection systems (e.g. liners and leachate treatment systems),
- protection of the desired waste-stabilisation processes within the landfill,
- protection against human-health hazards.
- Examples of waste-property based criteria are:
- requirements on knowledge of total composition,
- limitations on the amount of organic matter in the waste,
- requirements or limitations on the biodegradability of the organic waste components,
- limitations on the amount of specified, potentially harmful/hazardous components (in relation to the above mentioned protection criteria),
- limitations on the potential and expected leachability of specified, potentially harmful/hazardous components (in relation to the above mentioned protection criteria),
- ecotoxicological properties of the waste and the resulting leachate.

The property-based criteria for acceptance of waste must generally be most extensive for inert waste landfills and can be less extensive for non-hazardous waste landfills and least extensive for hazardous waste landfills owing to the higher environmental protection level of the latter two."

The actual criteria as given in Council Decision 2003/33/EC (see section 2.2) were subsequently developed by a Technical Adaptation Committee (TAC). Due to the very tight time schedule for this work, only protection of groundwater was taken into account. Protection of surface water (fresh or marine) was not considered. Similarly, there was little or no specific consideration of protection of the environmental protection systems or protection of the desired waste stabilisation processes within the landfill. It could, of course, be argued that criteria that would protect the groundwater would also, to a large extent, protect both the environmental protection systems and the desired waste stabilisation processes. Surface waters, on the other hand, may both be more and less vulnerable than groundwater to impacts from leachate, depending on the character of the surface water body and the quality (and quantity) of the leachate.

Annex I to the Landfill Directive (1999/31/EC) lists the general requirements for all classes of landfills, including requirements on geological barriers and top and bottom liners. Section 3.4 in Annex I specifies certain conditions under which the requirements on barriers and liners may be reduced as mentioned below.

2.2 EU acceptance criteria

As indicated above, the EU Landfill Directive 1999/31/EC entered into force in 1999. Three types of landfill were defined in the directive: landfills for “inert waste”, for “non-hazardous waste”, and for “hazardous waste”. The Council Decision establishing criteria and procedures for the acceptance of waste at landfills pursuant to the Landfill Directive was completed in December 2002, and published as Decision 2003/33/EC. The Council Decision lays down criteria for wastes to be accepted at landfills for inert waste, landfills for hazardous waste and for stable, non-reactive hazardous waste to be accepted at non-hazardous waste landfills as well as for non-hazardous waste disposed in the same cell for non-hazardous waste as stable, non-reactive hazardous waste.

The acceptance criteria given are primarily limit values for the release (leaching) of predominantly inorganic contaminants from the waste and, to a limited extent, maximum compositional values for some other contaminants (e.g. the content of organic carbon). The Landfill Directive has been adopted in accordance with Article 130 S (now Article 175 (1)) in the EEC Treaty. This means that each Member State has the opportunity to implement more stringent criteria than those listed in Council Decision 2003/33/EC if it can be justified as necessary in order to protect environment. The Council Decision does not include specifications (methods, criteria) for all categories of landfills and contaminants of importance. Criteria for other sub-categories of landfills may be developed on a national basis. For example, the criteria for acceptability of non-hazardous (optional) and solidified and stabilized (monolithic) waste are to be set at a national level. Member states may also set additional criteria for contaminants that are not included in the Council Decision.

The Council Decision allows member states the possibility - on a case-by-case basis - to apply limit values that have been increased by up to a factor of three for specific contaminants. However, the use of higher limit values requires a risk assessment to demonstrate that there will be no additional risk to the environment, e.g. because of additional measures have been taken to reduce the release from a landfill. Annex I to the Landfill Directive allows reductions in the requirements for a geological barrier and an artificial liner provided that an assessment of the environmental impact shows that the landfill poses no unacceptable risk to soil, groundwater or surface water.

The methodology used to set the waste acceptance criteria in terms of leaching properties is outlined in section 4.2 and described in more detail in Appendix 6.

A summary of the EU criteria for acceptance of waste at landfills is presented in Appendix 3.

2.3 Classification of waste

The Council Decision is linked to the waste classification (hazardous/non-hazardous). The links between waste classification and landfill categories are illustrated in Figure 1. Wastes must be disposed according to their classification if no exception is mentioned (as it is, for example, for stable, non-reactive hazardous waste).

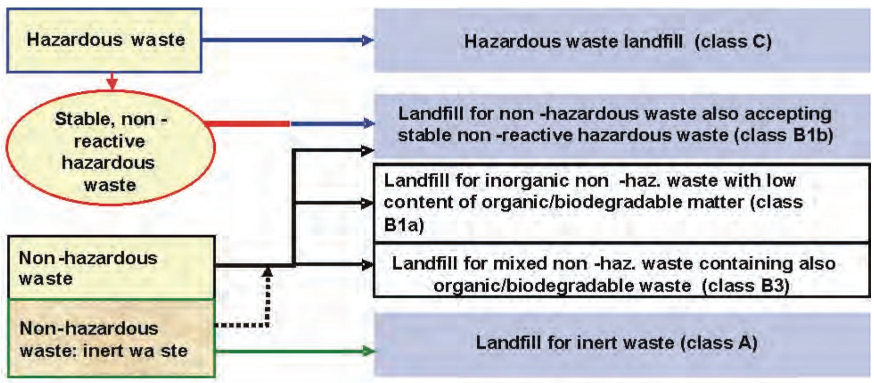

Figure 1. Links between waste classifications and landfill categories. EU criteria given for landfill categories marked with blue colour. (Wahlström et al 2006)

The Council Decision provides acceptance criteria for:

- Waste to be disposed of in a landfill for inert waste;
- Hazardous waste to be disposed of in a landfill for hazardous waste;
- Stable, non-reactive hazardous waste to be disposed of in a landfill for non-hazardous inorganic waste; and
- Non-hazardous waste to be disposed of in the same cell as stable non-reactive hazardous inorganic waste.

Inert waste is defined in the Landfill Directive, Article 2 (e) as a special type of non-hazardous waste. Only inert waste which complies with the waste acceptance criteria (WAC) for inert waste landfills may be received at a landfill for inert waste. Inert waste could, in fact, also be accepted at landfills for non-hazardous waste.

Hazardous waste is defined by the Hazardous Waste Directive (91/689/EEC) and the European Waste Catalogue (Council Decision 2000/532/EC as amended.) It is a waste that can cause special danger to health and to the environment because of its chemical or other property. In practice, the classification is usually based on the total content of hazardous substances. Stable, non-reactive hazardous waste that meets the WAC for waste to be landfilled in landfills for non-hazardous waste receiving stable, non-reactive hazardous waste can be disposed placed in this specific category of landfills. The acceptability of waste for landfill must always be determined. So the waste acceptance criteria have no relationship to the hazardousness of these wastes and there may be wastes that are deemed to classified as hazardous and fail the WAC for hazardous waste landfills and therefore cannot be landfilled. Such waste must be further treated to meet the criteria or it must be landfilled underground.

Waste that is not defined as hazardous is classified as non-hazardous. Non-hazardous waste is not, however, automatically approved for acceptance at a landfill for non-hazardous waste. In general, the acceptability of such waste for landfilling must be evaluated through a basic characterisation. At EU level the basic characterisation of non-hazardous waste (other than that mentioned above) does not include a testing requirement. Such requirements may be implemented by the member states at national level for sub-categories of non-hazardous waste landfills.

2.4 Priority substances

The following directives and regulations have clear connections to impact evaluation:

- In the Water Framework Directive (Directive 2000/60/EC 33), 33 priority substances or groups of substances posing a risk to waters have been listed (Appendix 4). The vast majority (29) of these 33 priority substances are organic, with polycyclic aromatic hydrocarbons (PAH) as a significant part (3 specific PAHs plus PAH as a group). Among the 33 substances are 3 metals and their compounds (cadmium, mercury and lead) defined as priority hazardous substances (PHS), and nickel and its compounds are subject to review for identification as possible PHS. Moreover, arsenic is identified in Annex VIII of the Water Framework Directive as a substance liable to cause pollution
- The old Groundwater Directive (1980/86/EEC) and directives relating to drinking water, the new Groundwater Directive 2006/118/ec and possibly Directive 2006/11/EC on pollution caused by certain dangerous substances discharged into the aquatic environment of the Community (Codified version). However, as the latter is a codified version it may contain no new legislation.
- The Waste Framework Directive
- The Persistent Organic Pollutants (POP) Regulation (850/2004): ban on landfilling of waste and limits for underground deposits of waste containing PCB above certain limits.
- The HELCOM about discharges to Baltic Sea. The Helsinki Commission, or HELCOM, works to protect the marine environment of the Baltic Sea from all sources of pollution through intergovernmental co-operation between Denmark, Estonia, the European Community, Finland, Germany, Latvia, Lithuania, Poland, Russia and Sweden.
- PARCOM (the Paris Convention) for the North Sea. the PARCOM Recommendation 88/2 on the Reduction in Inputs of Nutrients to the Paris Convention Area sets a target of 50%, compared to input levels in 1985, for reduction in inputs of nutrients into areas where these inputs are likely, directly or indirectly, to cause pollution.

2.5 Implementation of The council decision in the Nordic countries

Council Decision 2003/33/EC describes the procedures for acceptance of waste to landfill. At national level, decisions are needed on test procedures, selection of L/S for evaluation of acceptance criteria, limits for PAH content in inert waste and WAC for landfilling of monolithic waste. Moreover, guidance is needed on the performance of basic characterisation. The implementation of the Council Decision at a national level will to a certain extent be based on landfill strategy, waste types and disposal areas available in each country.

Table 1. Examples of implementation of the Council Decision in Nordic legislation.

Country	Examples of national decisions	Remarks
Denmark	EU acceptance criteria to be evaluated at L/S = 2 l/kg or, if that is not possible, at L/S = 10 l/kg, using percolation and/or batch tests	Landfills to be situated near the coast No use of impermeable mineral layer in top cover
Finland	In principle, all wastes except municipal wastes, asbestos, wastes on inert lists and a few specified wastes require testing EU acceptance criteria are evaluated at L/S 10 Recommendation for leaching test methods (percolation test for significant waste streams and inert wastes, for others primarily two-stage batch test) Guidelines given for interpretation of test data	Top cover required for non-hazardous & hazardous waste landfills
Norway	All waste except non-hazardous waste that is deposited in a landfill for non-hazardous waste is to be tested prior to landfilling Acceptance criteria shall be evaluated at L/S = 10 l/kg and L/S = 0.1 l/kg (C_0).	
Sweden	All waste except non-hazardous waste that is deposited at a landfill for only non-hazardous waste is to be tested prior landfilling. There are transitional rules for existing landfills. In some cases testing is not necessary for those landfills until the end of 2008. Acceptance criteria to be evaluated at L/S = 10 l/kg and Co (L/S = 0,1 l/kg) For regularly produced waste more advanced testing which includes percolation testing is required. For other waste streams and compliance testing the two stage batch test can be sufficient.	Top cover required for non-hazardous & hazardous landfill

2.6 Other related national regulation

An assessment of the environmental impact of leached components is also needed for industrial by-products and wastes to be used in engineering works. In some Nordic countries, national regulation for the use of certain material in construction projects already exists or is under preparation.

The Danish Statutory Order no. 655 of June 27, 2000 on Recycling of Residual Products and Soil in Building and Construction Work specifies the criteria for utilisation of contaminated soil and waste materials in terms of total content and leachability of various contaminants (at L/S = 2 l/kg).

The Finnish Ministry of the Environment has recently published a Government Decree 591/2006), which covers the use of selected environmentally compliant waste materials in construction projects. In cases where limit values are exceeded or materials are not within the scope of the regulation, an impact assessment may be performed to demonstrate a possibly acceptable risk from waste utilisation in construction works.

In Sweden, criteria for use of waste materials in construction and on top cover (the growth layer) are under preparation.

3. Considerations prior to deviation evaluation

3.1 Risks related to landfill design and operation

Examples of potential risks associated with the acceptance wastes with higher acceptance values are shown in Table 2. The risks are mostly waste and substance specific, but may also depend on the landfill design.

Table 2. Overview of potential risks that may be caused by deviations from EU criteria in a landfill scenario.

Exposure target	Impact	Example
Disposed waste	Loss of stability with time (influence on aftercare) Influence of leachate on other disposed waste materials	Loss of total solids (e.g. salts)
Landfill construction layer	Change in physical properties (e.g. increased water permeability) Breakdown of construction layers	Ion exchange with mineral layer (e.g. bentonite) Corrosion (sulphate)
Landfill leachate treatment	Influence on treatment process difficult to meet discharge limits	Nitrification process

The actual risks related to the landfill design depend particularly on the source strength and the attenuation rate of specific substances in the landfill leachate. Some of the important parameters influencing the release from a landfill (the source strength) are:

- the height of the landfill;
- the rate of infiltration and percolation of (rain)water through the top cover and leachate through the bottom liner;
- the length of the period during which leachate is collected, including the lifetime expectancy of the bottom sealing and the top cover;
- the leaching characteristics of the waste.

The three first parameters affects the development of the liquid to solid ratio (L/S) in the landfill with time and hence the concentration of the leachate when it reaches the unsaturated zone below the landfill. They are, in principle, independent of the leaching properties of the waste. The latter parameter, the leaching characteristics of the waste, is of course waste-specific.

The influence of each of these and other parameters on the outcome of an assessment of the environmental impact of a landfill is discussed in more detail in section 5.5.

3.2 Assessment of alternative waste management options

Alternative waste treatment and management options for waste not meeting the EU landfill acceptance criteria need to be evaluated before the possibilities for deviation from the EU criteria are discussed. The evaluation needs to cover the total environmental burden from the management options including treatment and management of all rejects from the processes considered (i.e. waste water, air emissions, solid waste streams). Also economical, technical and political issues (e.g. the ban of certain wastes) need to be taken into account.

Life-cycle analysis (LCA) provides a tool for the calculation and comparison of environment impacts from different waste treatment options. The setting of the boundary for the life-cycle analysis is an important question, e.g. to determine if the study is limited only to waste treatment options or if it should also cover the process from where the waste arises. The assumptions made for the assessment need to be well documented and transparent because the results are strongly dependent on such assumptions.

LCA is a comparative method. The outcome of an analysis is usually reported in terms of energy requirements or emissions per ton of waste material, taking into account all incoming and outgoing material streams, energy inputs and discharges from the system. The most difficult part of the life-cycle assessment is the evaluation of the different impacts.

Figure 2 shows an example of the scope of an LCA study on the treatment of a solid waste incineration (MSWI) residue which contains environmentally hazardous substances and which cannot be landfilled at landfills above ground without additional treatment or without deviation from the EU landfill criteria. By treating the residue, its characteristics and leaching properties should be improved. However at the same time, the treatment process consumes resources and energy and causes emissions. When evaluating the overall benefits and disadvantages of residue treatment, both the quality of the treated residue and the emissions of the treatment process should be taken into account. Additionally impacts related to the treatment of leachate at the landfill also need to be taken into account.

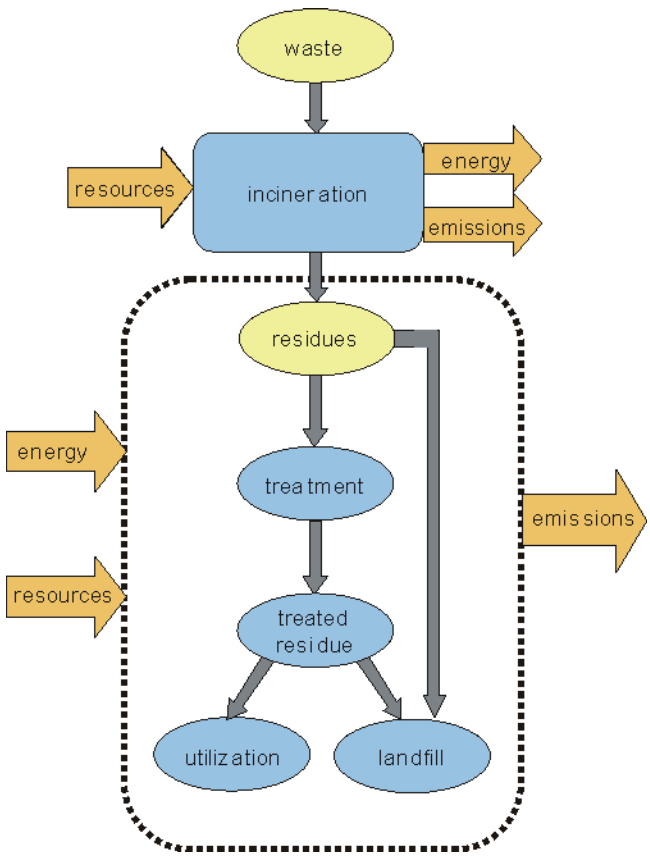

Figure 1. The Scope of an LCA study. Example: treatment of APC-residue.

3.3 Review of the classification of waste

It is important to check that the classification of the waste has been done correctly. Cases have appeared where waste producers have classified a waste as hazardous “just in case”. Another typical example that may cause incorrect classification is the use of old classification data or common practice.

The re-classification of a waste stream is usually a very demanding and is a complex task, especially when the judgement is based on the assessment of hazard H14 “Ecotoxicity”. In general, it must be shown that those H-properties which originally were used to classify the waste as hazardous have been changed to indicate non-hazardousness. The evaluation is often carried out on a case-by-case basis or by local authorities where local waste management policy may play a role. However, classification can also be seen as national policy issue.

NOTE: The classification of waste as hazardous or non-hazardous is not part of this study.

4. Tools for risk assessment for deviation evaluation

4.1 General aspects in risk assessment

Guidelines for the risk assessment of contaminated sites have been developed and can in many cases form the basis for an evaluation of environmental risks. These guidelines are often very broad and general, covering numerous exposure pathways to humans and the environment. In particular, the future use of a contaminated site is considered in the risk assessment. When carrying out an assessment of the environmental impact or risk from a landfill, it will, in many cases, not be necessary to consider as many exposure pathways.

Risk characterisation means listing and describing, and if possible quantifying, all the unfavourable properties or conditions related to a particular scenario, e.g. as doses or emissions and their impacts. The aim of a risk assessment is to determine for a particular waste management option the conditions under which the undesired phenomena are minimised or the conditions under which they can perhaps be totally eliminated. This generally includes following steps:

1. Identification of dangerous substances (hazard identification)
2. Estimation of emissions to environment and risks for human health (exposure assessment)
3. Risk assessment (risk estimation)
4. Sensitivity analysis due to variations of input data (risk characterisation). – Steps 3 and 4 are often linked in an iterative process.
5. Final conclusions

Some typical steps involved in the estimation of environmental risk are shown in more detail in table 3. Note! Table does not cover estimation of risks to human health because these risks generally not considered critical in this context.

Impacts from a particular substance are often linked to a specific scenario. An important part of the risk assessment is to evaluate the impact of critical parameters on the results. Numerous uncertainties are involved in the use of scenarios, site descriptions, coupled models, and assumed or averaged parameters and data. Experience has shown that the assumptions made concerning the water balance (resulting from a particular design and operation of the application) as well as the release mechanism from the product have a particularly strong influence on the results.

The following properties are generally of concern in an environmental risk assessment:

- the direct toxicity of the waste to plants, animals, and water organisms;
- the leaching properties with special attention paid to all relevant conditions in specific scenarios (as a consequence of the transport of harmful substances to the environment)
- the formation of toxic gas emissions, e.g. due to volatilisation, degradation, reactivity
- dust problems
- the risk of ignition (fire) and the consequent emissions
- The following exposure pathways, though, are considered relevant with regard to landfilling and also to utilisation of waste materials:
- the influence on plants and animal species in direct or indirect contact with the waste;
- the influence on the groundwater and surface water quality during the whole lifecycle (storage, transport, use, disposal).

Table 3. Typical steps in estimation of environmental risk. The risk assessment can be simplified in specific cases taking into account only relevant risk or steps.

Step	Information needed, tools available	Needs, challenges
General	Overall scheme	Development of framework for risk assessment Guidance document including sampling etc with reference to EN standards
Description of scenario	Selection of substances of concern Information on material properties Information on characteristics of disposal place (design, operation, location, hydrology, stability etc) Identification of harmful properties during the whole life cycle of the waste including the most important handling steps	Guidelines for justification of selected scenario and input data
Identification of relevant exposure routes	Descriptions of process and operational conditions Risk management measures	
Evaluation of emission doses to environmental compartments and exposure of critical targets	Transport by water: Choice of appropriate leaching test. This means that the mobility of the substance from the product r and also the factors controlling the leaching need to be assessed usually through a full characterisation of the leaching behaviour. Suitable test methods for the assessment are now under preparation in the European standardisation organisation CEN (CEN TC 292). Use of transport models. Several transport models available Transport by air: Estimation of emissions to air (including dust emissions)	Development of guidelines for modelling and selection of input-data (model includes several assumptions). Special attention needs to be paid to the differences between the conditions of the real case and of the model presented. Information for the environmental assessment of organic pollutants taking into account their special properties (degradation, colloid formation) Validation studies needed

Table 3. continued Typical steps in estimation of environmental risk.

Step	Information needed, tools available	Needs, challenges
Risk estimation	Calculation of critical doses for target groups The assessment of ecotoxicological risks can only be made roughly, because only limited suitable reference values for no-observed effect concentrations (NOECs), predicted no-effect concentration (PNEC) or lowest observed effect level (LOEC) are available.	Guidance on evaluation of exotoxicological risks (methods, reference values) The reference values are usually derived for certain laboratory conditions and certain chemical compounds. The applicability of these values may be poor and therefore they should be used with caution. The ecological effects on plant and animal species are evaluated by comparing the calculated values with the toxicological values reported in literature
Accidents	Especially concerning transport (soil & groundwater pollution), fires (soil, groundwater, air)	
Uncertainty analysis	Variations in input data and checking the influence of critical assumptions give important information on the sensitivity of the results. Examples of typical parameters to be checked are: Influence of product properties (e.g. density, permeability, thickness, chemical composition and leaching behaviour) on doses Dilution factor in the environment for substances in the leachate, i.e. assumptions in the calculation of water flow (rate, percolation, surface wash)	Validation of the model work.
Conclusions	Good documentation including relevant input data and assumption. The conclusions and the chosen acceptance criteria need to be carefully explained	Development of guidance document for proper and transparent documentation

4.2 Methodology for impact assessment

4.2.1 The approach in the EU modelling work

In the definition of inert waste in Council Directive (1999/33/EC, Article 2), it is stated that the risk from leachate (from a landfill) must be insignificant, and in particular not endanger the quality of surface water and/or groundwater. In the development of the waste acceptance criteria the interpretation of insignificant risk was linked to the selection of an acceptable risk level. After discussions among experts, drinking water standards were chosen as the basis for an acceptable risk level in affected groundwater, mostly because no groundwater quality criteria, which would have been more relevant, existed at international/EU level. Due to higher toxicity of some metals (Cu and Zn) to aquatic life than to humans, a few adjustments were made to the drinking water standards which are based on human health (see Appendix 5: Quality criteria used in development of EU criteria).

The basis for the development of the actual criteria was the outcome of modelling work done by several EU experts on selected landfill scenarios (Hjelmar et al., 2001 and 2005). The resulting groundwater quality was modelled at two distances from the edge of the landfill as well as at the surface of the groundwater directly below the landfill. The leaching behaviour of the waste and the corresponding leachate quality were simulated in these scenarios. One of the scenarios was a landfill for inert waste where no protective measurements, e.g. no top sealing or leachate collection and treatment systems were in place. Also landfills for non-hazardous and hazardous waste landfill were modelled, taking into account the requirements outlined in the directive for the top and bottom sealing layers and the collection and treatment of leachate. Here, only the leaching and transport of inorganic components were modelled. Based on the modelling results, the maximum leaching values in the leaching test corresponding to a specified groundwater quality could be determined.

These results were subsequently modified politically by the Commission and the TAC, taking into account specific waste characteristics for certain landfill categories. Especially values for arsenic, copper, lead, antimony, mercury and selenium were increased for hazardous waste landfills (see Table 5 in Section 5).

The starting point in landfill modelling is the identification of the nature of the risk (contaminant source and exposure risk). The prerequisite is a reliable knowledge of waste characteristics e.g. composition and leaching behaviour. In practice, the modelling work is linked to a specific scenario in which material properties are known or anticipated and the exposure conditions defined. It should also be noted that the procedure involves numerous simplifications and generalisations of complex and diverse physico-chemical processes. The sensitivity of the results to these assumptions should always be checked.

The approach used in estimating the impact of a landfill or a landfilled waste material on downstream groundwater (or surface water) quality may be illustrated by Figure 3, which shows three models coupled in series.

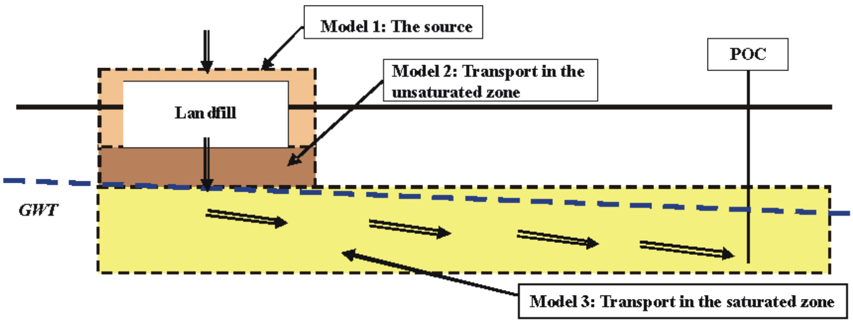

Figure 2. Cross-section showing the principle of using three coupled source and transport models for the calculation/estimation of the impact of a landfill on downstream groundwater quality (at the point of compliance, POC). From Hjelmar et al. (2005).

Transport modelling work may best be described in terms of a series of consecutive steps. The background for TAC work is presented in details in Appendix 6. The stepwise procedure is outlined below (see also Figure 4):

1. choice of primary target(s) and principles
2. choice of critical parameters and values
3. choice and specification of landfill scenario
4. choice and specification of environment scenario
5. description of the source of potential contamination
6. description of the migration of the contaminants from the landfill to the points of compliance (POCs)
7. evaluation of the impact (as a function of time) at the POCs
8. sensitivity analysis

The modelling is often an iterative process, where the influence of changing conditions and matrix ageing are estimated. Critical and safe disposal scenarios causing both high and low releases into the environment are identified, when possible. Besides modelling the release from waste to groundwater and surface water also some prediction of the long-term behaviour of the waste is important for each scenario modelled.

4.2.2 General requirements for appropriate modelling

It should be stressed that models only show an estimate of the release from waste under certain assumptions. A scenario may be used to compare impacts under different conditions and should not be taken as an absolute estimate without caution and without considerable refinement. The modelling results reflect the quality of the input data and the assumptions made, and may easily give a result that is either too favourable or too unfavourable compared to reality.

The following general requirements are important and should be addressed in modelling work:

- a qualitative systematic description of the landfill and groundwater system and the design of a conceptual model
- identification of the main critical exposure routes via a source-pathway-receptor analysis
- a definition of accepted risk/reference (e.g. that drinking water requirements should be met at a certain point of compliance)
- a description of the questions to be answered and a listing of input data needed for the calculation (e.g. influence of the water permeability of the landfill top layer on concentrations at POC)
- selection of an appropriate (mathematical) model. The complexity of the model must be sufficient to address the vulnerability of the site and consistent with the quality of the input data. In many situations a simple model may perform as well as a more sophisticated model, depending on the quality of the input data and the uncertainty in the conceptual model. Transparency is easily lost when using models that are too complex. When using complex numerical models, bench mark calculations (preferably analytical solutions) should be carried out to facilitate the evaluation of the results produced by a complex model.
- consideration of the sensitivity of modelling results (influence of assumptions)
- knowledge of the restrictions imposed by waste management legislation (especially in the case of hazardous waste)
- the requirements of end-users (are results easily understandable?, ”weaknesses and strengths” of the modelling work)
- the competence of the modeller with sufficient background, model knowledge, references etc.
- a list of issues specific to the site in question: how are the background concentrations in the surrounding environment considered in the modelling?

Figure 4 shows an example of a risk assessment procedure and the expected results.

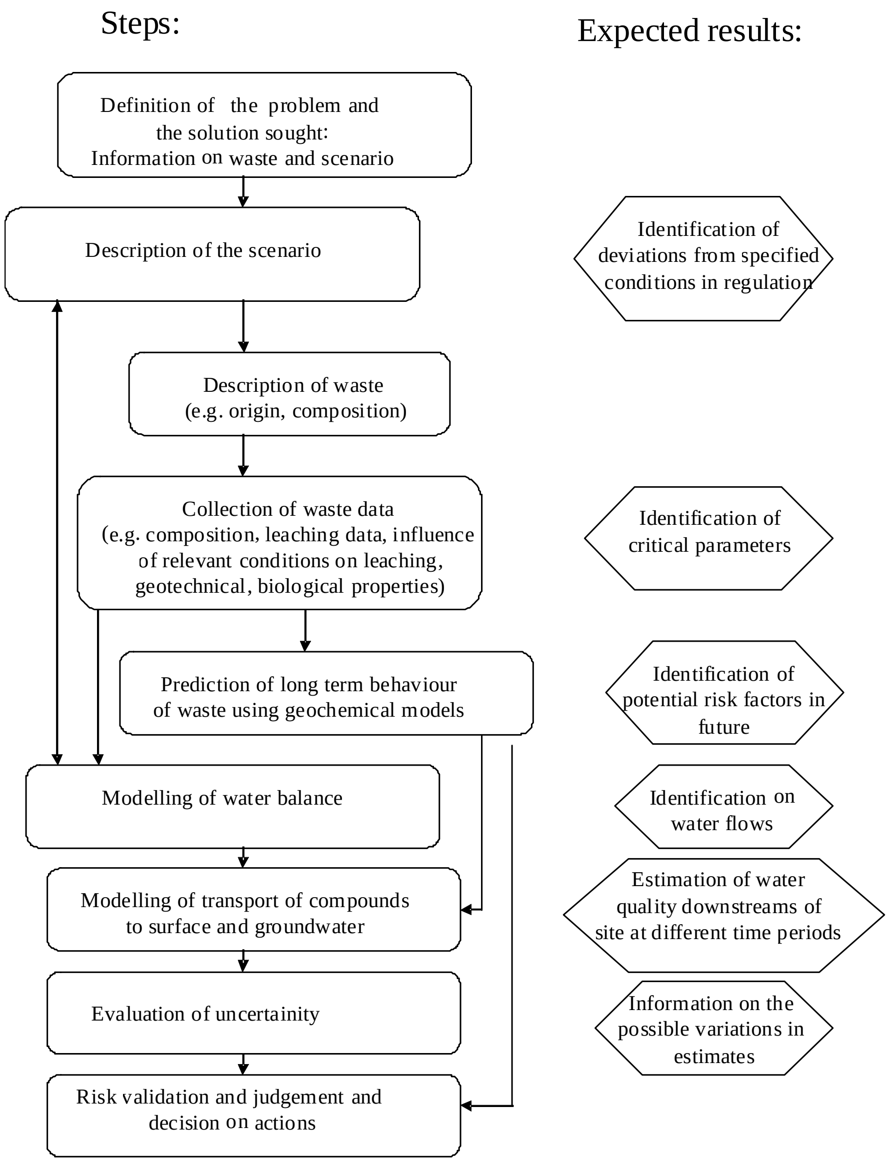

Figure 3. Risk assessment procedure (here limited to transport of contaminants with water). Case: deviation in landfill construction.

4.3 Use of ecotoxicity tests for supplementary information

Besides traditional leaching methods, ecotoxicity tests can be used to provide supplementary information on waste properties. Assessment of biological effects brings forth material properties which cannot necessarily be judged directly from concentration data (e.g. due to unknown bioavailability, interactions of compounds in biota, the presence of unknown organic contaminants). Biotests developed for solid waste characterisation can be performed either on eluates prepared from the waste or directly on the waste material itself. Usually a test battery consisting of several different tests is needed due to the specific sensitivities of different test species in the single biotests. Only biotests performed on water samples are discussed in this context.

The results of a biotest generally depend not only on the sensitivity of the test species in the case of concern; they are also strongly dependent on the actual L/S ratio resulting from the preparation of the test medium, and particularly on the mode of eluate separation during this preparation. In many European countries an L/S ratio of 10 l/kg has been used for evaluation of waste acceptance and also for waste classification.

The results of a biotest express the percentage concentration of an eluate in a test medium corresponding to a certain (e.g. limited) toxicity or inhibitory effect (e.g. in terms of immobility or growth inhibition) or a non-toxic effect on the test species. In the tests, results are often reported in terms of the effective concentration EC50 value (which means the concentration of the test eluate, which causes specified effects on 50 % of the test species population) or the NOEC (which means the no observed effect concentration of the eluate).Test results from biotests can also be expressed as Toxic Units (TU). The toxic units (TU) can be calculated by multiplying the inverse of the effective concentration (EC) by 100 (TU = 100/EC50). Several approaches are used for the evaluation of Toxic Units. One approach suggested for wastes (Vaajasaari, 2005) is the following:

TU< 2	Not toxic
2< TU< 10	Toxic
10<TU <100	Clearly toxic
TU > 100	Significantly toxic

Table 4 shows biotests performed on eluates prepared from waste materials that have been selected in the European standardisation organisation CEN (CEN/TC 292 “Characterisation of waste”) for an inter-laboratory study on the precision in the performance of biotests.

In a Swedish research project (Anon. 2008) on use of biotests for waste classification, the suitability of following biotests is evaluated:

- Microtox® with bacteria *Vibro fisheri* (EC50 after 30 minutes)
- Growth inhibition test with algae *Pseudokirchneriella subcapitata* (primary producer; EC50 after 72 hours)
- Larval development test with the harpacticoid copepod *Nitocra spinipes* (primary consumer NOEC/LOEC after 6 days)
- Fish embryo test with zebrafish, *Danio rerio* (secondary consumer; NOEC/LOEC after 48 hours)

The reasons for selection of these methods are low sensitivity to salt, pH and also reasonable short test durations. This project gives valuable information on the selection of biotest batteries for wastes.

Table 4 Biotests selected by CEN/TC 292 for an inter-calibration study.

Bioassay	Test parameter and test time	Expression of results (examples)	Remark
Microtox Bioassays with the bacterium *Vibrio fischeri (*ISO 11348-3)	Luminescence 30 min	EC20- and EC50-values	Quick and easy test for general toxicity. However, applicability of test very selective (often not very sensitive for pollutants)
Daphnia magna (ISO 6341:1996E)	Immobility 24 h	ECX (e.g. EC50) and NOEC values	Sensitive to inorganic and organic compounds
Algae *Pseudokirchneriella subcapitata* (ISO 8692)	Growth rate 72 h	ECX (e.g. EC50) and NOEC values	Sensitive to heavy metals (e.g. copper)

EC= effective concentration
NOEC= no observed effect concentration

5 Application of the factor 3 rule

5.1 Needs and challenges

5.1.1 Identified needs in the Nordic countries

The need for higher acceptance values in Nordic countries was discussed in a workshop arranged in October 2007 (see Appendices 1 & 2). Some typical wastes for which the application of the factor 3 rule could be considered are:

- Municipal solid waste incinerator air pollution control (MSWI APC) residues (chlorides)
- various fly ashes
- metal dust
- metallurgical sludge
- aluminium waste (electrolysis waste)
- shredder waste (DOC, PCB)
- possibly textile waste
- contaminated soil

Some of this information was obtained from a Swedish study (SGI report 555). There will be future ban on the landfilling of shredder waste in Finland. All MSWI APC residues/fly ashes in Denmark are currently exported, primarily to Langøya in Norway (NOAH).

Some of the potential risks to landfill operation involved in allowing landfilling of waste with high salt contents were discussed. These may include corrosion problems, loss of mass and mechanical stability, influence on clay liner properties and influence on the leachate treatment process.

The criteria for acceptance of waste at landfills for non-hazardous waste landfills in the Nordic countries were also discussed. The situation was reported as follows:

- Denmark: criteria will be developed for non-hazardous, mineral waste (a DOC problem may exist for soil).

- Finland: recommendations for leaching criteria are given (however, very lightly contaminated soils are often accepted on the basis of total concentrations of contaminants and used for daily cover of landfills)
- Norway: non-hazardous waste landfills are regulated through permits.
- Sweden: no testing requirements for waste to non-hazardous waste landfills (except those accepting stable, non-reactive hazardous waste)

In impact assessments and criteria setting, antimony (Sb) and selenium (Se) often create problems with compliance due to the very low drinking water quality criteria.

The following other related needs have also been identified (here the focus is on non-hazardous waste materials):

- Deviation in landfill construction (bottom liner, top liner) especially for wastes from energy production
- Deviation of acceptance criteria for wastes to be disposed in landfills situated in areas with various levels of environmental vulnerability. This would include cases where the disposal conditions differ significantly from the conditions considered when the WAC were set, for example in cases of sensitive disposal environments or in cases where the construction consists of several different by-products. It might also be possible through impact evaluation to accept higher emission values in cases where more information is available about the behaviour of the harmful components in the selected scenario.
- Links to utilisation of waste in soil construction work and landfill construction work
- Wastes to be landfilled, where national criteria may be set for contaminants that are not included in the EU regulation.

5.1.2 Specific issues related to the interpretation of leaching test data

The impact assessment (i.e. estimation of resulting concentrations in surface/groundwater at a selected POC) required to justify the application of the factor 3 rule will generally be based on results from leaching tests. When using leaching test results, certain issues may need consideration. Such issues may include:

- How to interpret laboratory leaching data on a waste material (e.g. the influence of crushing of monolithic material). This should be considered when using waste-specific information (see section 5.5.4),
- The long term behaviour of waste and the effect of mixing of wastes. This is part of risk assessment and will generally not be covered by the results of a laboratory leaching test. Long term behaviour may in some cases be estimated by supplementary hydrogeo-chemical modelling.

- Evaluation of results if, for example, the release at L/S = 2 l/kg exceeds the criteria but the release at L/S = 10 l/kg is ok. This should be resolved at a national level.

5.1.3 Link between calculated and actual waste acceptance criteria

In the TAC modelling group, scenario-based model calculations were carried out and leaching criteria referring to specific L/S values were developed as described in section 4.2.1. However, the calculated “risk based” leaching criteria (“WAC”, waste acceptance criteria) were subsequently subject to political negotiations between the members of the TAC. In this process, the preferences and needs of the individual EU member states were discussed, and the effect of changes of various WAC values was investigated using a large database containing leaching properties of a number of common waste types. The difference between the calculated WAC and the WAC that were finally adopted and implemented into Council Decision 2003/33/EC may be illustrated by table 5, where WAC calculated by DHI are compared to the WAC from the Council Decision. It can be seen that for non-hazardous and hazardous waste landfills, the actual EU WAC (with the exception of the criteria for Cr) may be up to 20 times higher than the values determined by the risk-related procedure. For inert waste landfills with no required top covers or bottom liners, most of the actual WAC are much closer to or even lower than the calculated WAC.

The fact that the actual WAC differ from the calculated, risk-related criteria presents a potential problem or at least requires a decision in relation to the use of the factor 3 rule. If a site-specific risk or impact assessment is carried out to determine whether or not a 3 time increase of a leaching criterion for a specific component will result in maximum resulting concentration of that component at the POC, should the water quality criterion for the component at the POC then be recalculated to reflect the political change (if any) of the WAC, or should it remain at the value chosen for the TAC calculations or at an appropriately chosen local value (as far as groundwater is concerned)? It could be mentioned that the vast majority of the participants in the workshop (see section 5.1.1 and Appendix 2) preferred the latter solution.

Table 5. Comparison between modelled WAC (waste acceptance criteria) and the actual WAC set by the TAC in terms of leaching limit values (mg/kg) at L/S = 10 l/kg.

Parameter	Inert waste landfill			Non-hazardous waste landfill receiving stable, non-reactive hazardous waste			Hazardous waste landfill		
	Calcu-lated WAC	Actual WAC	Ratio*)	Calcu-lated WAC	Actual WAC	Ratio*)	Calcu-lated WAC	Actual WAC	Ratio*)
As	0.27	0.5	1.9	1.8	2	1.1	3.3	25	7.6
Ba	8.2	20	2.4	50	100	2.0	63	300	4.8
Cd	0.079*	0.04	0.51	0.57*	1	1.8	1.5*	5	3.3
Cr	3.7	0.5	0.14	27	10	0.37	78	70	0.90
Cu	0.86	2	2.3	5.9	50	8.5	14	100	7.1
Hg	0.016*	0.01	0.63	0.095*	0.2	2.1	0.1*	2	20
Mo	0.94	0.5	0.53	6.4	10	1.6	15	30	2.0
Ni	0.87	0.4	0.46	6.3	10	1.6	18	40	2.2
Pb	0.44	0.5	1.1	3.2	10	3.1	8.8	50	5.7
Sb	0.074	0.06	0.81	0.46	0.7	1.5	0.66	5	7.6
Se	0.093	0.1	1.1	0.62	0.5	0.81	1.2	7	5.8
Zn	2.9	4	1.4	21	50	2.4	55	200	3.6
Chloride	980	800	0.82	5800	15000	2.6	6000	25000	4.2
Fluoride	15	10	0.67	90	150	1.7	120	500	4.2
Sulphate	1600	1000	0.63	9200	20000	2.2	9300	50000	5.4
DOC	98	500	5.1	590	800	1.4	620	1000	1.6
Phenol index	3.6	1	0.28	-	-		-	-	

*) Ratio = actual WAC divided by calculated WAC

**) Determined at POC = 20 m and not at the groundwater below the landfill

5.2 Approaches in deviation impact modelling studies

5.2.1 Interpretation of the requirements for use of the factor 3 rule

Modelling is generally required to evaluate emissions and impacts, and if the evaluation produces a positive result, to demonstrate that the emissions (including leachate) from the landfill will present “no additional risk” to the environment, if the leaching WAC are increased by up to 3 times (as specified in Section 2 of the Annex to Council Decision 2003/33/EC). A “factor 3 risk assessment/model calculation” shall be carried out for specified wastes on a case-by-case basis.

To cause “no additional risk” could with some reason be interpreted as complying with the water quality criteria (WQC) at the POC. This could be taken literally as the WQC set by the TAC for the groundwater at the POC defined by the TAC, but it could also (and perhaps more sensibly) be based on local or national regulation and considerations. Groundwater-related WQC could hence be the same as or more restrictive than those defined for groundwater by the TAC, or they could address surface water quality, if that would be more appropriate in the specific situation. In the latter case, the POC would then be some point orarea in the surface water body, where the WQC (as influenced by the landfill) should be observed.

In some cases it might be possible to argue that groundwater quality criteria for certain components could exceed those used by the TAC.

5.2.2 Model improvements and actual changes

The model calculations could merely be performed in a more refined manner than the relatively crude "European average" calculations carried out by the TAC. By using site-specific data (e.g. describing landfill design and protective measures, the leaching properties of the waste in question, the climate and the water balance, the geology, the hydrology, the attenuation properties of the soil in the unsaturated zone and the aquifer, etc.) and possibly using more sophisticated release and transport models, a more "accurate" description of the resulting environmental impact adjusted to local conditions would result. Being more accurate does not imply whether or not the use of 3 x WAC will be justified; this would only be the case if the TAC calculations turn out to have been sufficiently conservative compared to the impact calculations performed with site-specific and waste-specific information.

The use of site-specific (and waste-specific) input data may be considered a requirement for the use of the factor 3 rule, and further sophistication of the models applied may be considered a general trend. While both may contribute to the production of better modelling results, it seems more interesting and appropriate in this context to study how actual changes in landfilling conditions in terms of location, design and operation of the landfill may influence the environmental impact of landfilling of a speci-fied waste at a chosen POC.

The following issues are important in relation to the site-specific impact assessment and in particular, the modelling of site-specific environmental impacts to determine if the factor 3 rule is applicable:

- Location of the landfill (possible relocation of the POC)
- The use of general or waste-specific input data in the source term
- The effects of changes in scenario conditions

These issues are discussed in more detail in the following sections.

5.2.3 Need for component-specific approach

At a national level, a discussion is needed on how to comply with the national policy in relation to prioritised substances (see 2.4). Here it is also necessary to consider that some of the WAC were elevated considerably compared to the calculated values as mentioned earlier in 5.1.3. For example, the limit value for Hg at hazardous waste landfills was increased 20 times. Due to the toxicity of both Cd and Hg, higher limit values for these substances are not generally recommended without special considerations. The strict approach for Cd and Hg is also in line with the current strategy of banning these metals in the Nordic countries.

On the other hand, some substances can be regarded as relatively non-toxic in certain environments (especially chloride). Also, more information on the toxicity and effects of some metals with very low limit values (e.g. Sb and Se) should be collated. If the outcome of this evaluation indicates that higher limit values could be justified in the Nordic countries (and elsewhere), a common approach for increases of WAC for these specific substances needs to be further discussed at the Nordic level and perhaps also at EU level.

5.3 Location of the landfill (possible relocation of POC)

Due mainly to time constraints the model calculations performed by the TAC were focused exclusively on the impact of the leachate from the landfill on the groundwater quality at the so-called point of compliance (POC), which was placed 20 m downstream of the edge of the landfill (for chloride and sulphate a POC placed 200 m downstream was accepted due to the relatively transient and non-toxic nature of the groundwater contamination caused by these two salts). As a consequence, the EU WAC set by the TAC are relevant to groundwater protection, but not necessarily to the protection of surface waster.

In many cases landfills in the Nordic countries (and elsewhere) are located near a surface water body and the leachate will, sooner or later, enter into this water body which may be a lake, a river or a marine water body (the sea). As described above, the effect on the surface water body is not accounted for in the calculations upon which the EU WAC are based.

The eventual discharge of the leachate into a surface water body may possibly provide some extra dilution which may (or may not) enable the application of the factor 3 rule. The main issues to be considered when this is determined are:

- the dilution potential of the surface water body;
- the definition and location of the POC(s);
- the WQC associated with the surface water body in question.

The WQC (and the POC) may be based on national or EU regulation or they may be locally determined. The POC may be a point in or it may be an area within the surface water body.

The easiest way to estimate the impact of a landfill on a POC in a surface water body will probably be to assume that the groundwater POC (according to the TAC calculations) is placed at the interface between the groundwater and the surface water. In that case, the relationship between the WAC and the WQC at the POC is known, at least in theory, and the additional dilution provided by the surface water body, including the initial dilution, may be estimated by hydrodynamic modelling, using local information on bathymetry, currents, wind, depths, dispersion, etc. In principle, this is just adding another model to the series of models described in section 4.2.1. Using site-specific information for the modelling of the unsaturated zone and groundwater transport parts of the scenario will, of course, also improve the accuracy of the results at the groundwater POC.

If the surface water quality criteria are stricter than the groundwater quality criteria applied at the groundwater POC, then some additional dilution will be required in the surface water body (or vice versa). When connecting the TAC groundwater POC and the associated influx of leachate into the groundwater with the surface water dilution model, it is important to remember that the TAC calculations were based on a landfill surface area of 40000 m^2 for non-hazardous and hazardous waste landfills. This means that if the landfill in question has a surface area that is larger than 40000 m^2, then the required surface water dilution based on the groundwater WQC will be increased proportionally (and decreased, if the surface area is smaller that 40000 m^2).

In the case of waste that exceeds the WAC for chloride and sulphate (e.g. MSWI fly ash), it should be fairly safe to apply the factor 3 rule to these two components for landfills located at or near a marine coast, since chloride and sulphate both are totally compatible with seawater, also at release concentrations corresponding to compliance with 3 x WAC. It will, however, be necessary to ensure either compliance with the WAC for other components, or acceptability of also applying the factor 3 rule to those other components that do not comply with the relevant WAC.

To change the distance from the downstream edge of the landfill to the POC may by characterised as a political issue; it does not change the environmental impact, it only changes the point where it is observed. There may, however, be good reasons to change the distance to the POC relative to the distances assumed by the TAC (see e.g. Appendix 5). Such a decision, which would have to be taken by the national authorities, could reflect the vulnerability and value of the downstream environment. It could be mentioned that a distance of 100 m (as opposed to the 20 m and 200 m used by the TAC) has been assumed in the Danish implementation calculations (Hjelmar et al., 2005). The effect of changing the distance to the POC is further discussed in section 5.5.

5.4 General or waste-specific source term

There are (at least) two different ways to apply a source term in the modelling:

- One way would be to use 3 x WAC (in this case C_0) combined with the appropriate parameter-specific value describing the release behaviour of a component (the so-called κ (kappa)-value, which is a measure of the rate with which the concentration level in the leachate decreases as a function of L/S) in the expression $E = (C_0/\kappa)(1-e^{-(L/S)\kappa})$, as input and evaluate the result at the POC. This would in principle mean that the evaluation is independent of the waste material under consideration. It should, however, be noted, that κ in reality is waste-dependent, but when the TAC calculations were carried out, the amount of data available on κ values was insufficient to justify a distinction between different waste types. The κ values used were determined for slightly polluted materials evaluated with respect to reuse, and are certainly not truly representative for all types of waste.
- Another way would be to use actual column leaching data for the specific waste material in question (provided the leaching does not exceed 3 x WAC) and again evaluate the result at the POC. In this case the evaluation would refer to a specific waste material.

The advantage of the first method would be the independence of the result on the waste material, which means that for scenarios and contaminants for which attenuation factors have been established, this attenuation factor may also be applied to other waste types. The advantage of the second method is that it more accurately describes the leaching behaviour of the waste in question. The results can, however, only be used for that specific type of waste. In principle, both methods should be applicable.

In a given situation, the most accurate results are likely to be obtained by the second method. This is illustrated in figure 5 which shows the source terms based on method 1 (based on κ for chloride by the TAC and 3 x C_0 for chloride at a landfill for hazardous waste) and on method 2 (based on actual column leaching test for chloride fly ash from a municipal solid waste incinerator (MSWI).

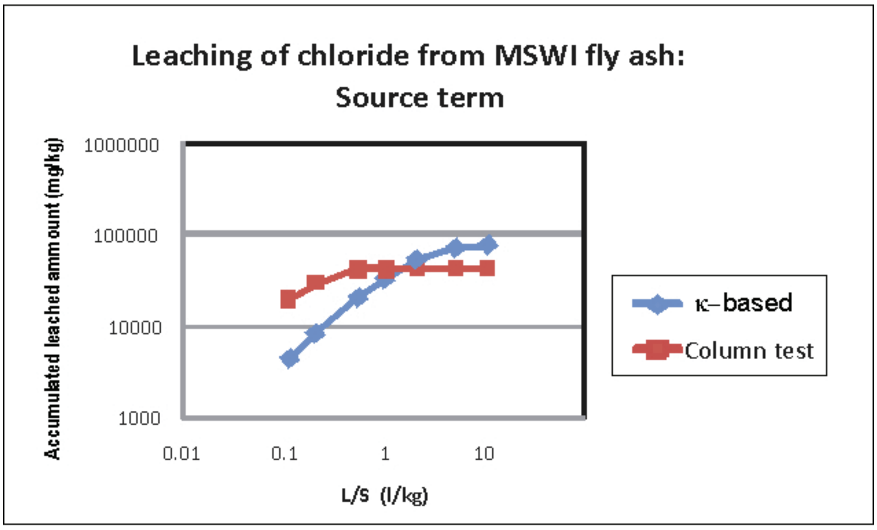

Figure 4. Source terms for chloride based on the equation described above using κ = 0.57 kg/l (as in the TAC) and 3 x C0 = 3 x 15000 mg/l (3 x as set for a landfill for hazardous waste by the TAC) and based on the results of a column test performed on MSWI fly ash.

As shown in figure 5, the source term for chloride will be underestimated at the beginning and overestimated later on if the general method 1 is used instead of the waste-specific method 2 in an evaluation of the applicability of the factor 3 rule in relation to acceptance of the MSWI fly ash at a landfill for hazardous waste.

Since the factor 3 rule can be applied only on a case-by-case basis anyway, it is recommended that waste-specific leaching data are used to describe the source term in accordance with method 2. This is further discussed in section 5.5.4 where a model impact calculation is carried out using waste-specific leaching data.

5.5 The effects of changes in various scenario conditions

5.5.1 The attenuation factor

In addition to getting more accurate modelling results from using waste-specific and site-specific data, the models may also be used to estimate the effects of actively introducing some changes to landfill design, landfill operation and landfill location relative to the conditions assumed in the TAC calculations or even relative to the conditions existing at a specific landfill. The modelling may show which types of changes would be most effective and provide estimates of whether or not solutions based on certain changes would be likely to reduce the impact of the leachate on the environment to a level that would make it acceptable to use 3 x WAC for some parameters and some wastes.

When considering the effect of various changes in the scenario conditions, it may be useful to express the results in terms of the attenuation factor, $1/f_a$, for each component under consideration. As described elsewhere, the attenuation factor is determined during "forward" calculation with the model and describes the ratio between the peak value, C_0, of that component in the leachate (in general assumed to occur when the first leachate is produced at the bottom of the landfill) and the highest value of the same component observed at any time later in the groundwater at the POC, i.e. $1/f_a = C_0/C_{POC\text{-}Peak}$. The attenuation factor is assumed to remain constant for a given component in a given scenario. When the competent authorities then fix a certain water quality criterion, WQC, for a given component in the groundwater at the POC, the corresponding value of C_0 for that component can be determined from the attenuation factor and the WQC by the equation C_0 = WQC x ($1/f_a$).

If it is assumed that the WAC to which the factor 3 should be applied is C_0, then the number to be compared to 3 x WAC is ($1/f_a$) x WQC for each component and each scenario. The relationship between the attenuation factor and the WAC at L/S = 2 l/kg and L/S = 10 l/kg is not totally linear (particularly if actual leaching data are used in the source term), but within the range of from WAC to 3 x WAC the relationship is rather close to linearity. This means that for all intents and purposes, an increase of a WAC by a factor of 3 will be acceptable, if a calculation based on a scenario principle comparable to the TAC scenario(s) and based on the same WQC at the POC as in the TAC calculations produces an attenuation factor equal to or higher than 3 times the attenuation factor found in the original TAC calculations.

Alternatively, the attenuation factor may be used together with the WQC to calculate C_0, which may again be used to calculate the corresponding values of "New WAC" at L/S = 2 l/kg and L/S = 10 l/kg using the equation shown e.g. in section 5.4. These "New WAC" can then be compared to the existing WAC, and if they are more than 3 times higher, then it has been shown for that component and that scenario, that an increase of the WAC by a factor of up to 3 will not cause any additional risk to the environment (as required in Council Decision 2003/33/EC).

In the following sections the effects of various changes in scenario conditions have been investigated, using modelling similar to that used in the TAC.

The measures studied include the following:

- Changes in the water balance
- Changes in the height of the landfill
- Changes in the distance to the POC (and the use of a waste-specific source term)
- Other changes

5.5.2 The effect of changes in the water balance

One of the scenario conditions that has the strongest influence on the attenuation factor resulting from the scenario calculations is the water balance, i.e. the rate of infiltration of precipitation through the top of the landfill and, in particular, the rate of release of leachate through the bottom liner. In a Danish study, three different scenarios were compared and the influence of the attenuation factor was evaluated. As discussed in section 5.5.1, the higher the attenuation factor, the higher the “permissible” WAC, and the higher the likelihood that the application of the factor 3 rule will be acceptable.

Scenarios studied:

- Sceanario 1 is selected as the baseline. Table 6 shows the water balance data used in the scenario calculations performed by DHI during the implementation of Council Decision 2003/33/EC in Denmark. In Denmark, WAC have been determined for three types of landfills: Landfills for inert waste, landfills for mineral waste (comparable to landfills for non-hazardous waste receiving stable, non-reactive hazardous waste) and landfills for hazardous waste. It is assumed that the bottom liner cease to function after 60, 80 and 100 years, respectively, for the three types of landfills. The rate of infiltration through the top is set to 350 mm/year for all three types of landfill throughout their lifetime. While it is functional, the efficiency of the bottom liner is estimated at 99 %.
- Scenario 2 in table 7 shows a change in the water balance for the landfills for mineral and hazardous waste, where it is now assumed that at the time when the bottom liner no longer is functional, a top cover is put in place reducing the rate of infiltration to 35 mm/year for mineral waste landfills and to 10 mm/year for hazardous waste landfills.
- In scenario 3 shown in table 8, the leaching of the waste in the landfills for mineral waste and hazardous waste has been accelerated (by doubling the rate of infiltration) during the periods when the bottom liners are intact. When they cease to function, the same top covers as in scenario 2 are applied.

Table 6. Scenario 1, which is comparable to the Danish implementation scenarios for Council Decision 2003/33/EC.

		Infiltration through the top			Leakage through bottom liner		
From	**To**	**Inert**	**Mineral**	**Haz**	**Inert**	**Mineral**	**Haz**
Year	Year	mm/year	mm/year	mm/year	mm/year	mm/year	mm/year
0	30	350	350	350	3.5	3.5	3.5
31	60	350	350	350	3.5	3.5	3.5
61	80	350	350	350	350	3.5	3.5
81	100	350	350	350	350	350	3.5
100	5000	350	350	350	350	350	350

Table 7 Scenario 2, where a top cover is placed on landfills for mineral and hazardous waste when the bottom liner cease to function.

From	To	Infiltration through the top			Leakage through bottom liner		
		Inert	Mineral	Haz	Inert	Mineral	Haz
Year	Year	mm/year	mm/year	mm/year	mm/year	mm/year	mm/year
0	30	350	350	350	3.5	3.5	3.5
31	60	350	350	350	3.5	3.5	3.5
61	80	350	350	350	350	3.5	3.5
81	100	350	**35**	350	350	35	3.5
100	5000	350	**35**	**10**	350	35	10

Table 8 Scenario 3, where accelerated leaching (doubled rate of infiltration) has been added to scenario 2 for landfills for mineral and hazardous waste.

From	To	Infiltration through the top			Leakage through bottom liner		
		Inert	Mineral	Haz	Inert	Mineral	Haz
Year	Year	mm/year	mm/year	mm/year	mm/år	mm/år	mm/år
0	30	350	**700**	**700**	3.5	7.0	7.0
31	60	350	**700**	**700**	3.5	7.0	7.0
61	80	350	**700**	**700**	350	7.0	7.0
81	100	350	**35**	**700**	350	35	7.0
100	5000	350	**35**	**10**	350	35	10

The attenuation factors resulting from scenario calculations using the water balance conditions for scenario 1, 2 and 3 as described above are shown in table 9. The remaining conditions (see e.g. Hjelmar et al. (2005)) were the same for all three scenarios.

The results in table 9 and table 10 clearly illustrate that changes in the water balance may have a dramatic effect on the attenuation factor and hence on the acceptability of applying 3 x WAC in specific cases.

Table 9 Attenuation factors resulting from scenario calculations based on the scenario conditions shown in tables 6, 7 and 8.

Parameter	Attenuation factor, 1/fa, for POC = 100 m						
	Scenario 1			Scenario 2		Scenario 3	
	Inert	Mineral	Haz	Mineral	Haz	Mineral	Haz
As	5.4	5.8	5.9	46	160	48	172
Ba	7.9	8.5	9.1	61	220	76	267
Cd	23	29	36	127	483	276	713
Cr tot	9.2	11	12	63	231	88	350
Cu	32	39	44	97	316	161	592
Hg	6.2	6.3	6.4	50	174	52	187
Mo	14	16	19	91	345	156	458
Ni	13	15	17	82	307	132	441
Pb	31	37	42	94	307	154	565
Sb	6.5	6.4	7.2	55	200	61	218
Se	11	12	15	92	333	121	364
Zn	13	14	16	81	300	127	429
Chloride	11	13	15	97	408	77	431
Fluoride	7.3	7.6	9.0	67	247	57	265
Sulphate	7.8	8.6	10	75	290	55	314
DOC	6.2	6.4	7.3	60	217	41	237

Table 10 Ratios between the attenuation factors found for scenarios 2 and 3 and for scenario 1.

Parameter	Ratio of attenuation factors for scenarios 2 and 3 relative to the attenuation factors for the baseline scenario 1			
	Scenario 2		Scenario 3	
	Mineral	Haz	Mineral	Haz
As	7.9	27	8.3	29
Ba	7.1	24	8.9	29
Cd	4.4	14	9.7	20
Cr tot	5.9	20	8.2	30
Cu	2.5	7	4.1	13
Hg	7.9	27	8.3	29
Mo	5.7	18	9.8	24
Ni	5.5	18	8.8	26
Pb	2.5	7	4.2	13
Sb	8.6	28	9.5	30
Se	7.6	22	10.0	24
Zn	5.6	18	8.8	26
Chloride	7.5	28	5.9	30
Fluoride	8.8	28	7.6	30
Sulphate	8.7	28	6.4	30
DOC	9	30	6.5	32

It should be noted, however, that the effect under Danish conditions is particularly strong because leakages of 350 mm/year are reduced to leakages of 35 mm/year for mineral waste and 10 mm/year for hazardous waste, which, by the way, would sacrifice some of the elements of sustainability associated with the Danish model. The WAC set by the TAC corresponds to a leakage that does not exceed 31.5 mm/year. The potential for increasing the attenuation factor by changing the water balance is therefore more modest for WAC based on the TAC calculations than for the case shown above (or it would require reducing the rate of infiltration to a very low level). It should also be noted that reducing the annual impact on the environment by reducing the rate of infiltration will prolong the period during which the cover must be effective.

The results also illustrates that the longer it is possible to collect the leachate, i.e. the longer the bottom liner and the leachate collection system remain intact, the higher a value of the attenuation factor may be obtained. It is, however, difficult to get a realistic estimate of the useful lifetime of bottom liners and leachate collection systems.

It may be concluded that changes in the water balance may have a substantial influence on the attenuation factor. The effect of reducing the leakage of leachate from 350 mm/year to 35 mm/year for mineral waste landfills and to 10 mm for landfills for hazardous waste ranges from increases of the attenuation factor from 2.5 to 9 times for mineral waste landfills and from 7 to 30 times for hazardous waste landfills, depending on the component. It should be noted however, that the leakage of leachate in the scenarios upon which the calculation of the EU WAC were based was 31.5 mm/year. The data do indicate that a reduction of the leakage from approximately 31.5 mm/year to 10 mm/year could increase the attenuation factor by approximately 3 times. It should also be noted that this would be likely to prolong the aftercare period substantially.

5.5.3 The effect of changes in landfill height

The height of a landfill may, as previously mentioned, have an effect on the attenuation factor and consequently on the acceptability of using 3 x WAC in a given situation. To illustrate this, the ratio between the attenuation factors estimated for a landfill measuring 5 m x 100 m x 100 m and the attenuation factors estimated for a landfill measuring 10 m x 100 m x 100 m has been calculated and shown in table 5.6. The estimates are based on scenario calculations (infiltration = 350 mm/year, no bottom liner, POC = 30 m downstream, 2 m unsaturated zone).

The ratios shown in table 11 compare well with previously reported changes of the attenuation factor of up to a factor of 2 for changes in landfill height from 10 m to 20 m. The actual change depends in particular on the kappa value associated with a given component.

It may be concluded that changing the landfill height will not have a major effect on the attenuation factor and is therefore not likely to be very useful in seeking the attenuation needed to make application of the factor 3 rule acceptable.

Table 11 Ratio between attenuation factors for landfills of 5 m and 10 m height.

Parameter	AF5 m/AF10 m
As	1.1
Ba	1.2
Cd	1.6
Cr tot	1.3
Cu	1.8
Hg	1.1
Mo	1.4
Ni	1.4
Pb	Not estimated
Sb	1.1
Se	1.2
Zn	1.4
Klorid	1.0
Fluorid	1.1
Sulfat	Not estimated
DOC	1.1

5.5.4 The effect of changes in the distance to the POC and use of waste specific data

Model calculations have been performed to investigate the effect of changing the distance to a downstream groundwater POC. The calculations were performed using data for a real waste, namely fly ash from municipal solid waste incineration (MSWI). Fly ash and dry/semidry acid gas cleaning residues from MSWIs generally have high contents of easily leachable chlorides and sometimes also sulphates and the leaching test results for these components often exceed the WAC for landfills for non-hazardous waste receiving stable, non-reactive hazardous waste and/or for landfills for hazardous waste. Whereas the leachable chloride content of semidry acid gas cleaning residues often exceeds even 3 x WAC, this is generally not the case for MSWI fly ash, which was therefore chosen for this exercise. Column leaching data for chloride and sulphate from the fly ash is shown in table 12. The relevant WAC from Council Decision 2003/33/EC are shown in table 13, and 3 x WAC is shown in table 14.

The model calculations have been carried out using the same conditions as used by the TAC, except that the actual leaching data shown in table 12 was used as source input instead of an analytical expression based on the CSTR model (see Appendix 6). This means that the resulting concentrations as a function of time at the POC can be compared directly to the GWQ used in the TAC calculations (= 250 mg/l both for chloride and sulphate). This is done in figure 6 for chloride and figure 7 for sulphate for POCs placed 30 m, 100 m, 200 m and 300 m downstream of the edge of the landfill. The somewhat odd-looking curves have two peaks due to the complicated water balance which exhibits substantial variations in the leakage during the first part of the first 80 years of the lifetime of the landfill (see e.g. Hjelmar et al., 2005). The calculations are the same for non-hazardous and hazardous waste landfills.

Table 12 Column leaching data (CEN/TS 14405) for chloride and sulphate on MSWI fly ash. The values for which European WAC have been defined (see table 12) are shown in bold.

Eluate fractions		**Chloride**		**Sulphate**	
From	**To**	**Concentration in eluate fraction**	**Accumulated leached amount**	**Concentration in eluate fraction**	**Accumulated leached amount**
l/kg	l/kg	mg/l	mg/kg	mg/l	mg/kg
0.0	0.11	**180000**	20000	**4100**	460
0.11	0.20	120000	31000	8700	1300
0.20	0.55	37000	43600	16000	6800
0.55	1.01	400	43700	16000	14000
1.01	2.12	140	**43900**	8800	**24000**
2.12	5.10	43	44000	1700	29000
5.10	10.57	15	**44100**	1800	**39000**

In table 13 the attenuation factors have been calculated for the POCs at different distances from the landfill. In table 14, using the attenuation factors from table 13 and the 3 x European WAC for chloride and sulphate, the resulting peak concentrations in the groundwater for POCs at the various distances from the landfill has been calculated for the scenario shown.

Table 13. 3 x European WAC for leaching of chloride and sulphate for landfills for non-hazardous waste receiving stable, non-reactive hazardous waste and landfills for hazardous waste.

Eluate fraction	Unit	WAC for chloride		WAC for sulphate	
		Non-haz*	Hazardous**	Non-haz*	Hazardous**
L/S = 0 – 0.1 l/kg***	mg/l	8500	15000	7000	17000
L/S = 0 – 2 l/kg	mg/kg	10000	17000	10000	25000
L/S = 2 – 10 l/kg	mg/kg	15000	25000	20000	50000

*: Landfills for non-hazardous waste receiving stable, non-reactive hazardous waste.
**: Landfills for hazardous waste.
***: Corresponding to C_0.

Table 14. European WAC for leaching of chloride and sulphate for landfills for non-hazardous waste receiving stable, non-reactive hazardous waste and landfills for hazardous waste.

Eluate fraction	Unit	3 xWAC for chloride		3 x WAC for sulphate	
		Non-haz*	Hazardous**	Non-haz*	Hazardous**
L/S = 0 – 0.1 l/kg***	mg/l	25500	45000	21000	51000
L/S = 0 – 2 l/kg	mg/kg	30000	51000	30000	75000
L/S = 2 – 10 l/kg	mg/kg	45000	75000	60000	150000

*: Landfills for non-hazardous waste receiving stable, non-reactive hazardous waste.
**: Landfills for hazardous waste.
***: Corresponding to 3 x WAC at C_0.

From table 15 it can be seen that under the conditions described, an increase in the attenuation factor of 1.7 for chloride and of 1.5 for sulphate may be achieved by moving POC from 30 m downstream to 300 m downstream of the landfill. As can be seen from table 16, this is not enough to reduce the calculated peak value at the POCs to the required 250 mg/l for both chloride and sulphate. At POC 200 m, the attenuation factor for chloride should be further increased by a factor of 2.8 (non-haz waste landfill) and a factor of 4.6 (haz waste landfill) to meet the groundwater quality criterion, and the attenuation factor for sulphate should be further increased by a factor of 6.1 (non-haz waste landfill) and a factor of 15 (haz waste landfill) to meet the groundwater criterion.

It might have been expected that an increase of the attenuation factor by a factor of 3 would be enough to ensure that the GWC is met at the POC when applying 3 x WAC. However, there are at least two reasons why this is not the case. The first reason is that the actual WAC for chloride and sulphate for non-hazardous and hazardous waste landfills were increased substantially relative to the calculated WAC (see table 5). The other reason is waste-specific: The release pattern of chloride and sulphate from the MSWI fly ash used in this example does not follow the ideal pattern described by the exponential decrease of the concentration as a function of C_0, L/S and the kappa values (= 0.57 kg/l for chloride and 0.33 kg/l for sulphate). In fact, the concentration of sulphate increases at low L/S values due to solubility control (see table 12).

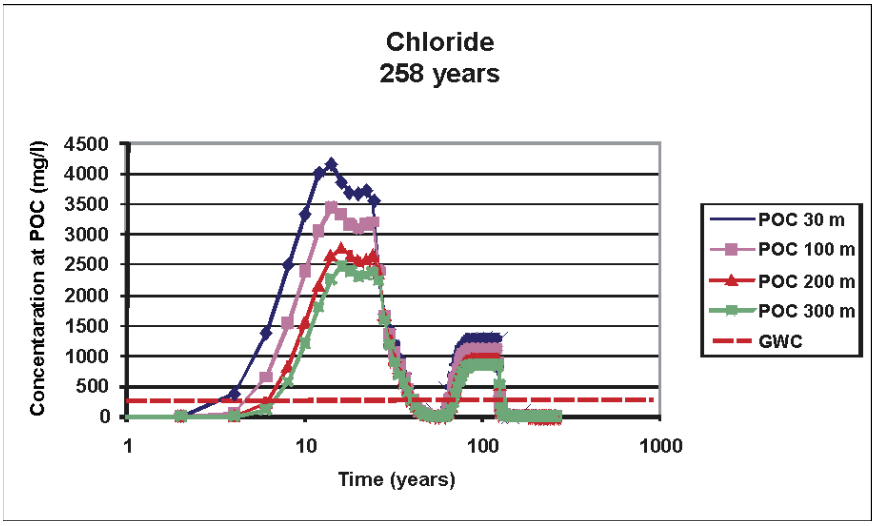

Figure 5. Results of model calculations for chloride using actual leaching data for MSWI fly ash as in-put.

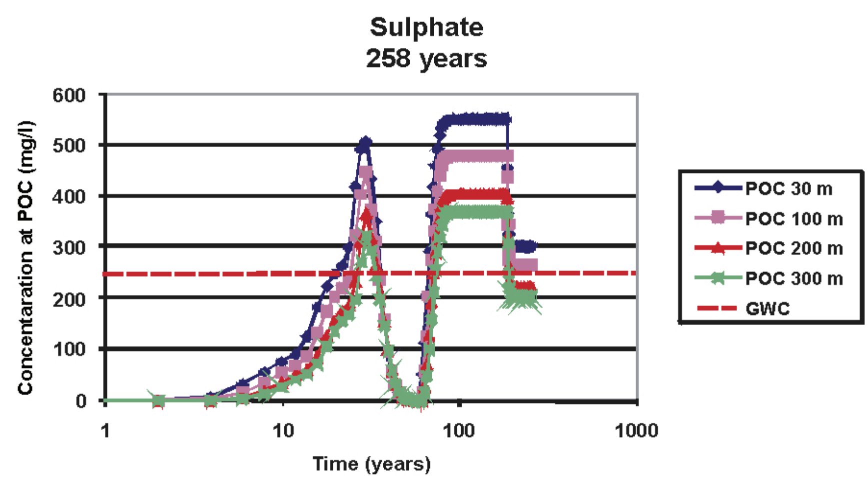

Figure 6. Results of model calculations for sulphate using actual leaching data for MSWI fly ash as input.

Table 15 Attenuation factors calculated from the scenario calculations using MSWI fly ash leaching data.

Distance from landfill to POC	Attenuation factor for chloride	Attenuation factor for sulphate
30 m	43	29
100 m	52	33
200 m	65	40
300 m	73	43

Table 16 Calculated peak values of chloride and sulphate in the groundwater for POC at various distances from the landfill when the source is based on 3 x WAC at L/S = 10 l/kg. The peak values should be compared to WQC of 250 mg/l for both chloride and sulphate.

Distance from landfill to POC	Peak value at POC for chloride (mg/l)		Peak value at POC for sulphate (mg/l)	
	Non-haz*	Hazardous**	Non-haz*	Hazardous**
30 m	1036	1726	2070	5175
100 m	864	1440	1804	4509
200 m	690	1150	1519	3797
300 m	620	1034	1384	3459

*: Landfills for non-hazardous waste receiving stable, non-reactive hazardous waste.
**: Landfills for hazardous waste.
WQC (groundwater quality criteria) for chloride: 250 mg/l
WQC (groundwater quality criteria) for sulphate: 250 mg/l

The use of waste-specific leaching data as opposed to a function based on kappa is generally recommended since it is likely to lead to a more accurate result of an impact assessment. It is, however, not possible to generalise as to whether or not the use of waste-specific data will lead to an impact that is higher or lower than the impact obtained with the kappa function in the source term and hence whether it will increase or decrease the chance of obtaining a result that will permit the use of the factor 3 rule.

5.5.5 The effect of other changes

Earlier sensitivity studies have indicated that if the groundwater velocity increases from the 20 m/year assumed in the TAC calculations to 100 m/year, the attenuation factor may increase by a factor of 2.

Different estimates of the value of the vertical dispersivity (a modelling parameter) in combination with the thickness of the aquifer may cause variations of the attenuation factor of up to a factor of 2.

5.6 Feasibility of using the factor 3 rule

Based on the previous sections, the following conclusions may be drawn in relation to the feasibility of applying the factor 3 rule:

If it is desired to landfill a waste material which exhibits a leaching behaviour that exceeds the WAC for landfills for non-hazardous waste receiving stable, non-reactive hazardous waste or for landfills for hazardous waste by less than a factor 3 for some components (not TOC, DOC, Cd or Hg) a further study may be of interest. Such a study could be conducted using an impact assessment model (or rather a series of models) similar in principle to that used by TAC when setting the WAC. It should be noted that the EU WAC for landfills for non-hazardous and hazardous waste originally were based on model calculations of the risk to downstream groundwater, but they were subsequently changed “politically” by a factor of up to 20 times before they were adopted.

The model should be used to determine the attenuation factors (i.e. the ratio between the maximum concentration at the base of the landfill and the peak concentration in the groundwater at the POC) for the components in question for scenarios of interest. Site-specific information should be used to the extent possible in the modelling, and the water quality criteria at the POC should reflect local conditions and regulation. If the landfill is located near a surface water body it may be relevant to include the dilution/attenuation in this surface water body and move the POC to the surface water body, using appropriate water quality criteria. It is recommended that waste-specific leaching data (column or lysimeter results) are used for the source term.

If using the site-specific and waste-specific information is not sufficient to increase the attenuation factor by a factor of at least 3 (depends on the local WQC), the modelling may be used to test the effects of potential measures that might be taken to increase the attenuation factor. Such measures could include:

- Changes in the water balance. Calculations indicate that a reduction of the leakage of leachate to 10 mm/year at the time when the bottom liner cease to function properly could increase the attenuation factor for most components by at least a factor of 3. The attenuation factor is more sensitive to changes in the water balance than to changes in most other conditions.
- Changes in the height of the landfill. Calculations indicate the increase in the attenuation factor caused by a reduction of landfill height from 20 m to 10 m would not exceed a factor of 2 and would be insignificant for several components.
- Changes in the distance to the POC. Calculations indicate that the increase in the attenuation factor obtained by moving the POC from 30 m downstream of the landfill to 300 m downstream of the landfill would be approximately 1.7 for chloride and 1.5 for sulphate.
- Changes in groundwater velocity and vertical dispersivity (these are site-specific condition that may be different from those assumed in the TAC calculations but they cannot be actively influenced). Earlier calculations have indicated that an increase in groundwater velocity from 20 m/year to 100 m/year could increase the attenuation factor by a factor of 2, and that changes in the estimates of the vertical dispersivity in combination with changes in the thickness of the aquifer could cause variations of the attenuation factor of up to a factor of 2.

If the application of the factor 3 rule is considered for a planned landfill rather than an existing landfill, it will be possible to a certain degree to choose a location with favourable attenuation properties.

A checklist for model-based evaluations of the feasibility of applying the factor 3 rule is provided in Appendix 7.

6. Summary and further work

6.1 Summary and conclusions

The EU landfill waste acceptance criteria (WAC) are mandatory minimum requirements for wastes to be disposed at specific landfills (i.e. landfill for inert waste, non-hazardous waste receiving stable, non-reactive hazardous waste or for landfills for hazardous waste). WAC for non-hazardous landfills not receiving stable, non-reactive hazardous wastes may be decided at a national level. The so-called factor 3 rule (allowing the WAC to be exceeded by up to a factor of 3) stipulated in Council Decision 2003/33/EC may be applied to the leaching criteria with restrictions for specific components in the waste (not dissolved organic carbon (DOC). The factor 3 rule cannot generally be applied for to other parameters (e.g. BTEX, loss on ignition). The only exception is the TOC content in certain special cases (e.g. soil-like waste to be disposed at landfills for inert waste). It is recommended not to apply the factor 3 rule to the WAC for priority pollutants such as Cd and Hg.

It should be noted that the factor 3 rule can only be considered in cases where the wastes exceed the WAC with less than a factor 3. Typical waste streams that may belong to this category are fly ashes from energy production, dust from process industry, metallurgical slag. A significant waste stream that often exceeds the leaching WAC by more than a factor of is APC residue from municipal waste incineration.

As specified in Section 2 of the Annex of the Council Decision, the use of the factor 3 rule requires a risk assessment to demonstrate that there will be no additional risk to the environment. Modelling is generally required as part of the impact evaluation of wastes with higher leaching characteristics than WAC and a “factor 3 risk assessment/model calculation” shall be carried out for specified wastes on a case-by-case basis.

Using the TAC modelling methodology (i.e. the modelling concept used in the development of the EU WAC criteria) is one way to evaluate the environmental impact but there are also other ways to assess this. There should be freedom of choice of method for impact evaluation, but the requirements to be fulfilled need to be fixed. It is recommended to interpret “no additional environmental impact” as meaning that the resulting effect at the POC should be fully acceptable under local conditions and fulfil local and national regulation.

In this study, the possibilities for deviation have been evaluated. Some of the main conclusions are:

- The use of site-specific and waste specific data may or may not be sufficient to allow an increase of the EU WAC.
- The location of the landfill and discharge of leachate into surface water body (e.g. sea, lake, marine water) can in some cases provide extra dilution which together with appropriate water quality criteria may (or may not) enable use of the factor 3 rule. This is particularly relevant to the release of salts (chloride and sulphate).
- In particular changes in the water balance at a landfill (i.e. the amount of infiltrating into and percolating through the waste and the functionality and expected lifetime of the bottom liner) may provide a change of the impact of leachate at the POC, which may justify the use of the factor 3 rule.
- The influence of several other landfill-specific data (e.g. height and distance to the POC) generally only has a minor influence on the impact at the POC and hence the acceptability of the use of the factor 3 rule.

Issues to be considered at a national level are discussed in the following section.

6.2 Issues to be decided at a national level

According to the Council Decision member states shall report to the Commission on the annual number of permits issued under factor 3-rule. In practice this means that when applying the factor 3-rule, there is a need for clarification of the approval system (i.e. by whom or at which level - e.g. local authorities, regional authorities - is the approval of increased waste acceptance criteria to be made). It should also be clarified who should be responsible for the submission of the request for increase of the waste acceptance criteria. Should it be the waste producer or the landfill operator?

Other aspect that need to be considered:

- if deviations only should be acceptable for certain landfill categories (it would seem reasonable not to allow the use of the factor 3 rule at landfills for inert waste which do not have any environmental protection systems)

- if there is a need for restrictions on parameters for which deviations are generally acceptable or not acceptable e.g. by reference to priority substances in the WFD
- selection of the distance of POC from the landfill and acceptance level for water quality at POC
- what should be included in the impact assessment (acceptable assumptions to be considered in landfill operation and constructions)
- identification of country-specific conditions that should be taken into account (both concerning country-specific waste materials and disposal conditions, e.g. relating to the groundwater table)

6.3 Reflections on Nordic challenges met during the implementation of the council decision

The following needs for further studies have been identified

- Collection of waste data (e.g. wastes not passing criteria due to low acceptance criteria for e.g. Se, Sb, F (inert waste)). Information is particularly needed on actual DOC levels in hazardous wastes .
- Guidance on how to take into account results from ANC (acid neutralisation capacity) determinations in evaluations of acceptance of waste at landfills.
- Information on how pH dependence test results are used in the Nordic countries.
- Knowledge about how waste-waste interactions in landfills can be taken into account.
- Collection of information on the specific Nordic situation relating to landfilling of waste (identification and analysis of problems, experiences in characterisation and testing, landfill conditions). Work aiming to result in a background report, e.g. to be used in discussions in EU (highlighting the current situations, experiences and problems in the Nordic countries).

References

Anon. 2008. Metodik för klassificering av H14-kriteriet i Avfallsförord-ningen. Förslag till biotestbatteri för klassificering av farligt avfall. Ekotoxikologisk testning med bakterie, alg, kräftdjur och fiskembryo. Avfall Sverige. Rapport 2008:16.

Chemical and Environmental Sampling - Quality through Accreditation, Certification and Industrial Standards", Radisson SAS Hotel Brussels, 14 – 15 April 2005. Arrangement of International Workshop in cooperation with Nordic Innovation Centre, CEN, Eurachem, Eurolab & VTT" on Proceedings. Edited by Komppa, V., Wahlström, M. & Laine-Ylijoki, J., 134 p.

Christensen, F., Bruijn, J., Hansen, B., Munn, S., Sokull-Klüttgen, B. & Pedersen, F. 2003) Assessment Tools under the New European Union Chemicals Policy. GMI 41. Spring 2003.

European Commission 2003. Technical Guidance Document for Risk Assessment. Part II. European Commission; joint Research Centre, EUR 20418 EN/2.

Harmonized methods for assessment of release of dangerous substances from construction materials. Funded by Nordtest. Project team: VTT, DHI (Denmark) and SGI (Sweden).

Hjelmar, O., H.A. van der Sloot, D. Guyonnet, R.P.J.J. Rietra, A. Brun & D. Hall (2001): Development of acceptance criteria for landfilling of waste: An approach based on impact modelling and scenario calculations. In: T.H. Christensen, R. Cossu and R. Stegmann (eds.): Sardinia 2001, Proceedings of the Eigth International Waste Management and Landfill Symposium, S. Margharita di Pula, Cagliari, CISA , Vol. III, pp. 712-721, CISA, Cagliari.

Hjelmar, O., Holm, J., Hansen, J.G. & Dahlstrøm, K. (2005) Implementation of the EU waste acceptance criteria for landfilling in Denmark. In: Proceedings of Sardinia 05, Tenth International Waste Management and Landfill Symposium, 3-7 October 2005, S. Margherita di Pula, Cagliari, Sardinia, Italy.

Vaajasaari, K: 2005. Leaching and Ecotoxicity Tests as Methods for Classification and Assessment of Environmental Hazard of Solid Wastes. Thesis. Technical University of Tampere. TUT Publication 540. 79 pages.

Wadstein, E., Håkansson, K., Tiberg, C. & Suèr, P. 2005. Kritiska deponiavfall – som inte klarar gränsvärden för att deponeras på deponi för farligt avfall. SGI Varia 555.

Wadstein, E., Rosqvist, H. & Frogner Kockum, P. 2006. Hälso- och miljörisker vid deponering av avfall som lakar höga halter av klorid och sulfat– En förstudie. SGI Varia 564.

Wahlström, M., Laine-Ylijoki, J., Hjelmar, O., Bendz, D. & Rosqvist, H. 2006. Models for impact evaluation on landfill – aspects for appropriate modelling. Nordic Innovation Centre. Technical report.

Wahlström, M., Eskola, P., Laine-Ylijoki, J., Leino-Forsman, H., Mäkelä, E., Olin, M. & Juvankoski, M. 1999. Risk assessment of industrial by-products used in earth construction (in Finnish). Espoo: Valtion teknillinen tutkimuskeskus. 79 p + app. 54p. (VTT Research Notes 1995).

Wahlström, M. , Laine-Ylijoki, J., Vestola, E., Vaajasaari, K. & Joutti, A. 2006. Acceptance procedures for waste at landfill. Environmental Administration Guidelines 2/2006. Ministry of the Environment. Edita Ltd. Helsinki 2006, 82 pages. (in Finnish).

Sammanfattning (Summary in Swedish)

Rådets beslut 2003/33/EG om kriterier och förfarande för mottagning av avfall bör implementeras i den nationella avfallslagstiftningen. Mottagningskriterier har givits för avfall som deponeras på

- deponier för inerta avfall
- deponier för miljöfarligt avfall
- deponier för icke farligt avfall som kan mottaga stabilt icke-reaktivt miljöfarligt avfall,

Den behöriga myndigheten kan från fall till fall ge tillstånd för avvikelser från EU:s mottagningskriterier:

- Gränsvärden för utlakning får dock höjas högst tre gånger ”faktor 3-regeln”, dock med undantag av DOC som inte får höjas)
- Gränsvärdet för totalt organiskt kol (TOC) får endast höjas två gånger

Gränsvärdet för andra parametrar (t.ex. BTEX, glödgningsförlust) får inte förhöjas tre gånger. Det enda undantaget utgör TOC-halten i vissa specialfall (t.ex. jordavfall som skall deponeras på deponier för inert avfall). Det bör också noteras att faktor 3-regeln endast kan tillämpas på avfall som inte överskrider mottagningskriterierna mer än tre gånger. Typiska avfallsströmmar inom denna kategori är flygaska från energiproduktionen, stoft från processindustrin och vissa metallurgisk slagg. En viktig avfallstyp som ofta överskrider mottagningskriterierna för utlakning mer än tre gånger är s.k. rökgasreningsaskor från sopförbränning.

Enligt rådets beslut får dispens endast medges under förutsättning att det är visat att ett högre gränsvärde inte medför någon ytterligare risk för människors hälsa eller miljön. Modellering är vanligtvis nödvändig som en del av riskbedömningen av avfall med hög utlakning och förutsätter en fall till fall studie med avfallets specifika egenskaper och deponeringsförhållanden.

En möjlighet är att använda TAC modelleringen (ett koncept som ligger som bakgrund vid utvecklingen av EG:s mottagningskriterier och som utarbetades av en expertgrupp under TAC), men också andra modelleringskoncept kan användas. Metoden för bedömning av miljöpåverkan kan väljas fritt, men specificering behövs om vissa antaganden gällanden deponin och accepterad risknivå (t.ex. vattenkvaliteten som skall uppfyllas i den utvalda referenspunkten POC). Det rekommenderas också att tolkningen av ”inte någon ytterligare risk” avser att halterna vid den utvalda referenspunkten (sk. point of compliance, POC) skall vara acceptabla under lokala förhållanden och uppfylla de lokala och nationella kraven.

I denna studie har dispensmöjligheterna utvärderats. Några av huvudslutsatserna är:

- Användning av plats- och avfallsspecifika data i modellering leder inte nödvändigtvis till att en höjning av kriterierna kan motiveras
- Deponins läge och lakvattnets utspädningsgrad (t.ex. till sjöar, havsvatten) medför i vissa fall att faktor 3 regeln kan tillämpas – närmast relevant för klorid och sulfat!
- Vattenbalansen i deponin (t.ex. infiltrationen genom sluttäckningskonstruktionen och bottentätningen) påverkar halterna i POC och kan därmed motivera en användning av faktor 3 regeln
- Deponikonstruktionen (t.ex. avfallsskiktets tjocklek) och avståndet till POC har vanligtvis en mindre betydelse för halterna i POC och möjliggör därmed knappast en förhöjning av mottagningskriterierna.
- Nationella beslut med anknytning till en förhöjning behövs bl.a. i följande:
- Klarhet om vilken instans besluter om en förhöjning (t.ex. lokal myndighet?). På vems initiativ skall förhöjningsmöjligheterna utredas (avfallsproducenten, verksamhetsutövaren för deponin)
- Kan en förhöjning tillämpas på alla typer av deponier (t.ex. deponier för inerta avfall, miljöfarliga avfall)
- Eventuella restriktioner för vissa ämnen (t.ex. Hg)
- Val av POC och vattenkvaliteten vid POC
- Vad skall inkluderas i en miljöbedömning (och därtill antaganden gällande konstruktion, deponeringsverksamhet)
- Eventuella landspecifika deponeringsaspekter som bör beakttas (t.ex. grundvattennivån)

Appendix 1: Workshop programme and participants

Risk assessment framework for management of the factor 3-rule in the Council Decision 2003/33/EC on waste acceptance criteria

DATE	**October 29-30, 2007**
PLACE	VTT, Vuorimiehentie 3, Otaniemi, Espoo, Finland
INTRODUCTION	The Nordic Landfill Group, under the Nordic Council of Ministers, has initiated a project entitled "Risk assessment framework for management of the factor 3-rule in the Council Decision 2003/33/EC on waste acceptance criteria" aiming to provide guidance on approach and key issues to be included in a risk assessment in cases where deviations in the Nordic landfill regulations are considered. The background of this project is the option given in the Council Decision to allow the use of up to three times higher limit values for specific specific components in wastes intended for landfill. However, the use of higher limit values requires a risk assessment to demonstrate that there will be no additional risks to the environment. There is today no guidance on the approach for such a risk assessment. Also the possibilities to deviate from the politically defined and in some cases increased limit values require discussions and background information. In relation to this a workshop will be arranged for invited Nordic stakeholders in October 29-30, 2007 in Finland. In the workshop topics related to the possibities for deviations and a draft report prepared by VTT and DHI will be discussed. The draft report will be sent to workshop participants prior to the workshop.
OBJECTIVES	To present and discuss a draft report, under construction by VTT and DHI, on a framework for risk assessment for management of the factor 3-rule in the Council Decision To get information on the needs of the Nordic national waste authorities. Discussions on difficulties and pitfalls in the interpretation of the outcome of risk assessments will be raised. National experiences with deviations will be discussed To get comments, e.g. from landfill operators, on effects of future demands in landfill operation in case of deviation of EU landfill criteria (e.g. practical problems, approval from authorities, costs) If possible, to agree on the framework of risk assessment required for deviations and discussion on elements in risk assessment to be decided on a national basis To open discussions on environmental protection level in landfilling of waste (definition needed? What is an environmental protection level of a landfill, are there any guidelines to be given?)
FINANCIER	Nordic Council of Ministers
ORGANISATION TEAM	VTT, Finland in cooperation with DHI –Water, Environment & Health , Denmark

Programme:

Chairman of the workshop: Jan Gronow (UK)

	Activity	Involved person
Oct 29, 1 pm	Welcome & Coffee	Cornelis Aart Meyles
	Introduction and presentation of participants	Margareta Wahlström & All
	BACKGROUND Definition of Environmental Protection level for landfill in EU legislation, UK approach in deviation	Jan Gronow (UK)
	Total risk concept: Use of life cycle analysis for emission evaluation	Jutta Laine-Ylijoki & Paula Eskola
	Break	
	TOPIC 1: CURRENT STATUS IN THE NORDIC COUNTRIES - NEEDS FOR DEVIATIONS	
	Short comments on current situations (landfill strategy) and needs for deviations (typical wastes and landfill scenarios for which deviations may be needed), demands on landfill operations in case of deviation, problems to be managed Discussion	Claes Ribbing
	Dinner	
Oct 30, 9 am	TOPIC 2: RESULTS AND EXPERIENCES FROM CASE STUDIES	
	Deviation using new site and material specific data, special case: discharge to sea	Ole Hjelmar
	Results from a Swedish study	Ebba Åkerlund
	Break	
	Short comments from EPAs & consults on tools for risk assessment (needs, challenges, elements to be included, experiences) Discussion on quality aspects in impact evaluations (requirements for proper modelling)	Steen Stentsøe
12-13	Lunch	
	TOPIC 3: FRAMEWORK AND APPROACH IN RISK ASSESSMENT	
	Presentation of the approach in the NMR draft report on factor 3-rule	Margareta Wahlström
	Short comments on the report Discussion on approach, identification of cases where deviation possible/impossible, issues to be decided on national basis	
	Coffee break	
	Discussion continues Recommendations, future actions, research needs	All
15.30	End of the workshop	

Workshop Participants:

Country	Participant	e-mail	Representing
UK	Jan Gronow	jan.gronow@btinternet.com	Lecturer
Sweden	Ebba Åkerlund	ebba.akerlund@swedgeo.se	Lecturer
Denmark	Ole Hjelmar	oh@dhigroup.com	DHI –water, environment & health, waste characterisation specialist . Member of project team
	Jonas Nedenskov	jn@av.dk	AV-Miljø, landfill operator
	Steen Stentsøe	sns@cowi.dk	COWI, landfill specialist
	Nils Jørgen Olsen	nijol@aar.mim.dk	Århus Environment Center, landfill inspector
	Jørgen G. Hansen	jogha@mst.dk	Miljöstyrelsen, Danish environmental protection agency
Finland	Ari Seppänen	ari.seppanen@ymparisto.fi	Ministry of the Environment
	Tarja Pinnioja-Saarinen	tarja.pinnioja-saarinen@rosknroll.fi	representing landfill operators (present only 29.10.2007)
	Martti Keppo	materra@elisanet.fi	landfill specialist
	Jan Österbacka	Jan.Osterbacka@ekokem.fi	expert in hazardous waste treatment
	Anna-Mari Lyytinen	anna-mari.lyytinen@ramboll.fi	Ramboll, consultant
	Pia Vilenius	pia.vilenius@chemind.fi	Association of Environmental Enterprises
	Jussi Reinikainen	jussi.reinikainen@ymparisto.fi	Finnish Environment Institute, expert on risk assessment of contaminated sites
Faroe Islands	Eyð Eidesgaard	eyde@hfs.fo	Food, veterany and environment agency
Iceland	Cornelis Aart Meyles	cees@ust.is	Environment and food agency
Norway	Gøsta Hagenlund	gh@fmsf.no	Authorities of County Governour in Sogn og Fjordane
	Ketil Haarstad	ketil.haarstad@bioforsk.no	Bioforsk, consultant http://www.bioforsk.no
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Organisation team	Margaret Wahlström	margareta.wahlstrom@vtt.fi	VTT, Technical research centre of Finland
	Jutta Laine-Ylijoki	jutta.laine-ylijoki@vtt.fi	VTT

Appendix 2: Report of discussions at the workshop

Risk assessment framework for management of the factor 3-rule in the Council Decision 2003/33/EC on waste acceptance criteria, held in October 29-30, 2007, Espoo, Finland

Topic 1: Current status in the Nordic countries – needs for deviations

It was suggested that the factor 3 rule should only be used if there are no other possibilities. Various treatment methods should be evaluated in view of technical feasibility, efficiency and economy prior to a decision to apply the factor 3 rule. LCC and LCA tools could be useful in obtaining an overall view of management options, including optimisation and use of resources. Transparency in such calculations is important. It is difficult to compare/weigh local and global effects, the evaluation must be site-specific.

Some participants were of the opinion that the potential application of the factor 3 rule should focus on major waste streams, others did not agree. It was mentioned that risk assessment is complex and demanding. It was also suggested that factor 3 – waste should be placed in separate cells in a landfill.

There is a need for special concern about potentially hazardous substances when/if applying the factor 3 rule. Special attention should perhaps be paid to substances such as Hg and to those substances for which the EU criteria were increased significantly in relation to the model results. The question of how to deal with waste for which acceptance at a landfill would require application of the factor 3 rule for several substances, was considered – but not resolved.

The group discussed whether or not hazardous waste after treatment can be classified as non-hazardous waste (e.g. MSWI fly ash/acid gas treatment residues) – this requires that the treatment to changes the properties upon which the hazardous classification was based. This is probably simplest for wastes with double entries in the European Waste Catalogue (EWC). Attention must also be paid to other relevant legislation and conventions, e.g. the POP regulation and HELCOM.

The group also discussed whether in the future there will be Nordic requirements/criteria for landfilling of non-hazardous waste (the implementation of the Council Decision in Denmark will include leaching limit values for mineral, non-hazardous waste). In the other Nordic countries no binding criteria are given.

The need for the application of the factor 3 rule to inert waste landfills was discussed. It was suggested that this would not be very desirable, since no environmental protection measures waste should be are required at such sites at the EU level (it may be at national level). The definition of inert taken into account if a risk assessment is carried out. It was mentioned that Denmark does not expect to have any inert waste landfills in the future.

It was pointed out that it may be possible to meet the requirements for application of the factor 3 rule where the hydrological conditions vary from the default scenarios. Variations in the characteristics of engineered liners of a non-hazardous landfill may also allow fulfil the requirements to be met..

Some typical wastes for which the application of the factor 3 rule could be considered are:

- MSWI APC residues (chlorides)
- various fly ashes
- metal dust
- metallurgical sludge
- aluminium waste (electrolysis waste)
- shredder waste (DOC, PCB)
- possibly textile waste
- contaminated soil from shooting ranges

Some of this information was obtained from a Swedish study (SGI report 555). There will be future ban on the landfilling of shredder waste in Finland. All MSWI APC residues/fly ashes in Denmark are currently exported to e.g. at Langøya in Norway.

Some of the potential risks to landfill operation involved in allowing landfilling of waste with high salt contents were discussed. These may include corrosion problems, loss of mass and mechanical stability, influence on clay liner properties and influence on the leachate treatment process.

Finally, the criteria for acceptance of waste at landfills for non-hazardous waste landfills in the Nordic countries were discussed. The situation was reported as follows:

- Denmark: criteria will be developed for non-hazardous, mineral waste (a soil DOC problem may exist).
- Finland: recommendations for leaching criteria are given (however, very lightly contaminated soils are often accepted on the basis of total concentrations of contaminants and, used for landfill daily cover. of landfills)
- Norway: regulated through permits.
- Sweden: no testing requirements for waste to non-hazardous waste landfills (except those accepting stable, non-reactive hazardous waste)

In impact assessments and criteria setting, antimony (Sb) often creates problems due to the very low drinking water criteria for Sb. Could the criteria for Sb be set too low?

Topic 2: Impact assessments: Modelling and results of/experience from case studies

A number of issues relating to the application of the factor 3 rule and the assessment of the resulting environmental impact have to be resolved or determined at a national level. Some of these may be:

- The exposure routes of concern
- The target and the acceptable risk level
- The distance to the point of compliance (POC)
- The need for case-by-case evaluation through modelling
- The need for national guidelines on the principles to be applied and on which parameters may be changed
- If there should be limits on the amount of “factor 3 waste” that can be placed in a landfill
- If there should be particular focus on short term or long term effects.

It was mentioned that impact assessment models already are used to obtain permits in several cases (mandatory in the UK). It was also mentioned that the WAC are fixed within the IPPC permit – which must be changed, if the factor 3 rule is applied at a later point in time.
There was agreement that the TAC modelling method is one way to evaluate the impact but there are also other ways to assess this. It is important thing is to create a reliable connection between the source and the impact at the receptor. There should be freedom of choice of method, but the requirements to be fulfilled should be fixed.
The input data to an impact assessment model will include results of leaching tests. If release rates or column leaching data are not available, a source function based on kappa values may be applied. The landfill scenario input will include the hydraulic conductivities of the liners, and the rate of infiltration may be varied. It was mentioned that in order to increase the rate of leaching and approach final storage quality as quickly as possible, top covers are not allowed in Denmark. However, it is currently being considered to require installation of a top cover of low permeability at the time of closure (or when the leachate collection systems cease to function) in order to be able to increase the calculated WAC for mineral waste and hazardous waste.
Based on calculations similar to those performed by the TAC, the following landfill design or operation changes will affect the ability to accept wastes containing contaminants with leaching values above the WAC without increasing the environmental impact at the POC:

- if the landfill height is 10m rather than 20 m, it may be possible to increase the acceptable concentration of contaminants by up to a factor of 2, depending on kappa;
- changes in the water balance at the site: may increase acceptable contaminant input concentrations by a factor of 10 to 20, depending on kappa and the efficiency and life expectancy of the top cover/bottom liner;

- changing the distance to the POC from 30 m to 100 m: may increase acceptable contaminant input concentrations by a factor of 1.2 of 1.7, depending on Kd/retention.
- if the groundwater velocity is 100 m/year rather than 20m/year: acceptable contaminant input concentrations may be increased by a factor of 2.

Topic 3: Other reflections

When considering the possibility of applying the factor 3 rule, the location of the landfill and the geological surroundings are important. Receptors other than groundwater may be relevant. In some Nordic countries surface water receptors are more relevant than groundwater receptors. In Denmark, new landfills will be placed near the sea, which in some cases provides up to 10000 times dilution at a marine POC downstream, near the coast. It is necessary to take the requirements of the Groundwater Directive and the Water Framework Directive into account. When limit values for e.g. surface water do not exist, it may be necessary to perform site-specific assessments for several parameters.

When the TAC calculations were performed, the results of the modelling were subjected to "political" discussion, and some of the final limit values in the Council Decision were up to 20 times higher than those resulting from the modelling (particular for non-hazardous and hazardous waste landfills). It seemed to be the general opinion that when determining whether or not a factor 3 may be applied in a given situation, it is not the politically determined WAC but rather the locally or nationally prescribed criteria at the chosen POC that should be observed (provided the 3 x WAC do not exceed 3 times the WAC in the Council Decision).

In the (TAC) calculations, it is assumed that the whole landfill contains only one type waste – and waste/waste interactions are ignored. If there is an upper limit on the amount of "factor 3 waste" allowed in a landfill, this could possibly be taken into account in the impact assessment/modelling.

It was mentioned that one should not forget the larger perspective – alternatives should be considered. There should be consideration of whether or not the waste could/should be pre-treated and whether the best solution is to collect and treat the leachate or to let it seep in a controlled way (saving resources, and eventually, the contaminants may end up at the same recipient).

The following question was asked: who should submit the request to increase the WAC by a factor 3? There was agreement that a site-specific request must be submitted by the landfill operator, possibly assisted by the producer of the waste in question (the producer must provide the characterisation data).

There was agreement that a clear conceptual description of the use of factor 3 and the associated requirements is needed.

Appendix 3: EU Acceptance criteria for landfills

Reference: The appendix of the EU´s landfill directive. Published in the EU´s official magazine L011, 16/01/2003 s. 0027 – 0049.

Table 1. Leaching criteria for two L/S ratios. Test method: percolation test prCEN/TS14405 or CEN-batch leaching test EN12457.

	Landfill for inert waste			**Landfill for non-hazardous waste landfill for inorganic waste (class B1b)** **Landfill for inorganic non-hazardous waste with a low content of organic/biodegradable matter**			**Landfill for hazardous waste**		
Parameter	C0 (mg/l)	L/S-ratio 2 (mg/kg)	L/S-ratio 10 (mg/kg)	C0 (mg/l)	L/S-ratio 2 (mg/kg)	L/S-ratio 10 (mg/kg)	C0 (mg/l)	L/S-ratio 2 (mg/kg)	L/S-ratio 10 (mg/kg)
Arsenic	0,06	0.1	0.5	0,3	0.4	2	3	6	25
Barium	4	7	20	20	30	100	60	100	300
Cadmium	0,02	0.03	0.04	0,3	0.6	1	1,7	3	5
Chrome (tot.)	0,1	0.2	0.5	2,5	4	10	15	25	70
Copper	0,6	0.9	2	30	25	50	60	50	100
Mercury	0,002	0.003	0.01	0,03	0.05	0.2	0,3	0.5	2
Molybdenium	0,2	0.3	0.5	3,5	5	10	10	20	30
Nickel	0,12	0.2	0.4	3	5	10	12	20	40
Lead	0,15	0.2	0.5	3	5	10	15	25	50
Antimony	0,1	0.02	0.06	0,15	0.2	0.7	1	2	5
Selen	0,04	0.06	0.1	0,2	0.3	0.5	3	4	7
Zinc	1,2	2	4	15	25	50	60	90	200
Chloride, Cl^-	460	550	800	8500	10000	15000	15000	17000	25000
Fluoridie, F^-	2,5	4	10	40	60	150	120	200	500
Sulphate, SO_4^{2-}	1500	560*	1000*	7000	10000	20000	17000	25000	50000
Phenol-index	0,3	0.47	1	-	-	-	-	-	-
Dissolved organic carbon, DOC#	160	240	500	250	380	800	320	480	1000
Total dissolved solid, TDS**	-	2500	4000	-	40000	60000	-	70000	100000

if the waste does not meet these values for DOC at its own pH value, it may alternatively be tested at L/S = 10 l/kg and a pH between 7,5 and 8,0. The waste may be considered as complying with the acceptance for DOC, if the result of this determination does not exceed 500 mg/kg.

*if the waste does not meet these values for sulphate, it may still be considered as complying with the acceptance criteria if the leaching does not exceed either of the following values: 1500 mg/l SO_4^{2-} as C_0 at L/S 10 l/kg (initial eluate from the percolation test) and 6000 mg/kg SO_4^{2-} at L/S =10 l/kg. It will be necessary to use a percolation test to determine the limit value at L/S = 0,1 l/kg under equilibrium conditions, whereas the value at L/S = 10 l/kg may be determined either by a batch leaching test or by a percolation test under conditions approaching local equilibrium.

**the values for total dissolved solids (TDS) can be used alternatively to the values for sulphate and chloride$^-$

Table 2: Other criteria.

	Landfill for inert waste	Landfill for non-hazardous waste landfill for inorganic waste (class B1b) Landfill for inorganic non-hazardous waste with a low content of organic/biodegradable matter	Landfill for hazardous waste
pH		>6	
Loss of ignition 550^{o}C			10 %*)
TOC	3 %**	5 % ***)	6 %***)
BTEX	6 mg/kg		
PCB	1 mg/kg		
Mineral oil (C10-C40)	500 mg/kg		
PAH-components	member state sets the limit		
Acid neutralization capacity (ANC)		to be evaluated	to be evaluated

*) Either TOC or LOI must be used for hazardous waste
** For soil higher limit value can be applied if DOC limit value is fulfilled (see Table 1)
***) For waste higher limit value can be applied if DOC limit value is fulfilled (see Table 1)

Appendix 4: Water framework directive

Table. List of priority substances in the field of water policy (Annex X of Decision N:o 2455/2001/EC)

CAS number	Substance	Identified as priority hazardous substance
15972-60-8	Alachlor	
120-12-7	Anthracene	(X) (***)
1912-24-9	Atrazine	(X) (***)
71-43-2	Benzene	
not applicable	Brominated diphenylethers (**)	(X) (****)
7440-43-9	Cadminum and its compounds	X
85535-84-8	C10-13-chloroalkanes (**)	X
470-90-6	Chlorfenvinphos	
2921-88-2	Chlorpyrifos	(X) (***)
107-06-2	1,2-Dichloroethane	
75-09-2	Dichloromethane	
117-81-7	*Di(2-ethylhexyl)phthalate (DEHP)*	*(X) (***)*
330-54-1	Diuron	(X) (***)
115-29-7	Endosulfan	(X) (***)
959-98-8	*(alpha-endosulfan)*	
206-44-0	Fluoranthene (*****)	
118-74-1	Hexachlorobenzene	X
87-68-3	Hexachlorobutadiene	X
608-73-1	Hexachlorocyclohexane	X
58-89-9	(gamma-isomer, Lindane)	
34123-59-6	Isoproturon	(X) (***)
7439-92-1	Lead and its compounds	(X) (***)
7439-97-6	Mercury and its compounds	X
91-20-3	Naphthalene	(X) (***)
7440-02-0	Nickel and its compounds	
25154-52-3	Nonylphenols	X
104-40-5	(4-(para)-nonylphenol	
1806-26-4	Octylphenols	(X) (***)
140-66-9	*(para-tert-octylphenol)*	
608-93-5	Pentachlorobenzene	X
87-86-5	Pentachlorophenol	(X) (***)
	Polyaromatic hydrocarbons	X
50-32-8	(Benzo(a)pyrene)	
205-99-2	(Benzo(b)fluoranthene)	
191-24-2	(Benzo(g,h,i)perylene)	
207-08-9	(Benzo(k)flouranthene)	
193-39-5	*(Indeno(1,2,3-cd)pyrene*	
122-34-9	Simazine	(X) (***)
688-73-3	Tributyltin compounds	X
36643-28-4	(Trybutyltin-cation)	
12002-48-1	Trichlorobenzenes	(X) (***)
120-82-1	(1,2,4-Trichlorobenzene)	
67-66-3	*Trichloromethane (Chloroform)*	
1582-09-8	Trifluralin	(X) (***)

(*) Where groups of substances have been selected, typical individual representatives are listed as indicative parameters (in brackets and without number). The establishment of controls will be targeted to these individual substances, without prejudicing the inclusion of other individual representatives, where appropriate.
(**) These groups of substances normally include a considerable number of individual compounds. At present, appropriate

indicative parameters cannot be given.
(***) This priority substance is subject to a review for identification as possible “priority hazardous substance”. The Commission will make a proposal to the European Parliament and Council for its final classification not later than 12 months after adoption of this list. The timetable laid down in Article 16 of Directive 2000/60/EC for the Commission's proposals of controls is not affected by this review.
(****) Only Pentabromobiphenylether (CAS-number 32534-81-9).
(*****) Fluoranthene is on the list as an indicator of other, more dangerous Polyaromatic Hydrocarbons.
(1) CAS: Chemical Abstract Services.

Appendix 5: Acceptance levels for POC in TAC work

Table. Development of EU criteria for waste acceptance. Acceptance levels chosen for different points of compliance (see section 3.4)

Parameter	K_d-values used in the TAC modelling (l/kg)	Acceptance level for groundwater quality at POC (mg/l)	Point of compliance (POC)		
			under bottom sealing	20 m from edge of landfill	200 m from edge of landfill
Sb	5	0,005		x	
As	50	0,01		x	
Ba	2	0,7		x	
Cd	20	0,005	x		
Cr	100	0,05		x	
Cu	14	0,05		x	
Hg	1	0,001	x		
Pb	50	0,01		x	
Mo	10	0,07		x	
Ni	50	0,02		x	
Zn	30	0,1		x	
Se	5	0,01		x	
F	2	1,5			x
SO_4^{2-}	0	250			x
Cl	0	250			x
DOC	0	10		x	

Appendix 6: Background information on TAC-approach

Part 1: The Step-wise Impact Assessment Procedure Applied to Groundwater Quality

(from Hjelmar, O.: Environmental performance of waste materials. In Dhir, R.K, Newlands, M.D. & Halliday, J.E. (eds.): Recycling and Reuse of Waste Materials. Proceedings of the International Symposium held at University of Dundee, Scotland, UK on 9-11 September 2003, Thomas Thelford, London, 2003, pp. 653-668)

Outline of the procedure

In this context, the procedure is used to set limit values for a waste to be used in a construction project. Only the impact on groundwater quality is considered. First a decision must be made concerning the primary target(s) or point(s) of compliance (POC), e.g. the downstream point(s) where the groundwater quality criteria must be fulfilled. Quality criteria are then selected for the groundwater and the physical characteristics of the construction project scenario and the environment scenario are selected and described. The environment scenario includes the net rate of infiltration and a hydrogeological description of the unsaturated and saturated (aquifer) zones upstream, below and downstream of the construction application. The source of the various contaminants is subsequently described in terms of the flux of contaminants as a function of time based on leaching data and the hydraulic scenario defined. Then the migration of the contaminants through the unsaturated zone into the groundwater and through the aquifer to the POC(s) is described with particular reference to the applicable K_d-values for each contaminant, which are used to calculate the retardation factors. The next step is to select and fit one or more models that can be used to describe the water flow and transport of contaminants from the base of the landfill through the unsaturated and saturated zones to the POC(s). The model calculations are carried out and "attenuation factors" (for granular waste the ratio between the source peak concentration and the peak concentration as modelled at the groundwater POC) are determined for each contaminant and POC. The attenuation factors are then used for a "backwards" calculation of the values of the source term corresponding to the selected groundwater quality criteria for each contaminant at a particular POC. The final step consists of transforming the resulting source term criteria to a limit value for a specific leaching test. The step-wise procedure is summarised below:

- Choice of primary target(s) and principles
- Choice of critical parameters and primary criteria values
- Description of the waste application scenario
- Description of the environment scenario
- Description of the source of potential contamination
- Description and modelling of the migration of the contaminants from the application to the POC(s)
- Performance of "forward" modelling to determine attenuation factors
- Application of the results to criteria setting ("backwards" calculation)
- Transformation of the source term criteria to limit values at different L/S values

Each step of the procedure is briefly discussed in the following. It should be noted that the procedure involves numerous simplifications and generalisations of complex and diverse physical-chemical processes. Only inorganic contaminants from largely inorganic waste materials are, for instance, considered. This is justified by the need to have an operational and relatively simple system, which can be used for the development of general criteria. Many of the technical details involved in this procedure are discussed in more detail in another paper in these proceedings [13] and in [11].

Step 1: Choice of primary target(s) and principles

The major potential impact during the service period of a waste application is believed to be migration of leachate and subsequent contamination of groundwater and possibly also of surface water and soil. Besides being contaminated itself, the groundwater will also be the potential conduit of a leachate plume to surface water bodies, and it is therefore convenient to express the primary environmental criteria in terms of a required groundwater quality. It is necessary to define the point(s) of compliance (POC), i.e. the location(s) where the groundwater quality must fulfil the quality criteria. This could potentially be in the unsaturated or saturated zone directly below the application or anywhere in the saturated zone downstream of the application.

Step 2: Choice of critical parameters and primary criteria values

It would seem appropriate to base the criteria aiming at the protection of groundwater on groundwater quality criteria. The latter are generally stricter than drinking water criteria since they take potential effects on the entire ecological system into consideration. Drinking water criteria only consider risks to humans consuming the water and, in addition, make allowance for substantial uptakes of e.g. Cu and Zn from water pipes. The problem is that whereas there are international criteria or guidelines (EU/WHO) for drinking water quality, no such international criteria exist for groundwater quality. In fact, national groundwater quality criteria exist only in very few of the EU Member States. The future implementation of the Water Framework Directive (see above) is likely to provide regional groundwater quality criteria within the EU. In the meantime existing drinking water criteria, possibly with modifications of some of those parameters, which are very high compared to normal groundwater values, could be used. The EU Drinking Water Directive [15] and WHO drinking water criteria [16] in combination provide limit values for the following inorganic components in drinking water: As, Al, B, Ba, Cd, Cr(total), Cu, Hg, Mn, Mo, Ni, Pb, Sb, Se, Zn, Br^-, Cl^-, CN^-, F^-, NH_4^+, NO_3^-, NO_2^- and SO_4^{2-}.

Step 3: Description of the waste application scenario

This step involves a detailed description of the physical appearance and properties of the waste application in question. Information should be provided on the shape, size, design, material types and geotechnical and hydrogeological properties of the application. The height, dry bulk density, length, width, surface area, volume, porosity, water content, permeability etc. of each separate layer of the application should be known. The information is used to determine or estimate the pattern of water flow through or around the waste material, the character of the waste material (granular or monolithic) and the likely leaching and transport mechanisms (equilibrium/convection or diffusion) controlling the release of contaminants. This forms the basis for the selection of appropriate leaching tests in step 5. Together with climatic information the scenario description is used to set up a water balance for the application. When the methodology is used for the setting of general limit values, the application scenario should be fairly “typical” and relatively simple. It should be noted that for some applications, e.g. waste materials used as sub-base or base materials in paved roads, it can be very difficult to describe the water flow within the road and the material. This is due to the combination of a low permeability cover, edge effects and unsaturated, preferential flow in cracks, and for the purpose of risk assessment, it may be necessary to apply a rather simplified flow description.

Step 4: Description of the environment scenario

The environmental scenario is intended to provide a simplified description of the characteristics of a specific (or in the case of limit value setting a “typical”) landscape in which the waste application scenario is placed, with particular focus on the hydrogeological properties of the area. The description should include climatic information (net rate of infiltration), information on the unsaturated zone (e.g. thickness, permeability, porosity, longitudinal dispersivity, bulk density, general geology) and the saturated zone (e.g. lateral flow velocity, aquifer thickness, porosity, longitudinal dispersivity, transversal dispersivity, bulk density, general geology, upstream groundwater quality).

Step 5: Description of the source of potential contamination

A description of the source of the contamination (the release of contaminants with the leachate) from the inert waste landfill is needed as an input to the groundwater transport/attenuation model. Since the source will change with time/progress of the leaching process, it is desirable to express the source as a function of time or, for percolation flow through granular waste, the liquid to solid ratio (L/S). It could be expressed as a time dependent flux in terms of unit mass of contaminant per unit mass of waste for percolation flow through granular waste, or in terms of unit mass of contaminant per unit surface area for monolithic waste materials and granular waste in a non-percolation situation.

In the following, only a simplified mineral granular waste flow-through system will be discussed. The granular waste under consideration may e.g. be MSWI bottom ash, crushed concrete, steel slag or coal fly ash. Continuing this simplification, it may be assumed that the waste application behaves similarly to a large column or lysimeter test carried out on the waste in question. An estimation of the flux of contaminants through the base of the site could then be based on the results of laboratory or lysimeter leaching tests combined with the hydraulic information on the landfill scenario obtained from step 3. The average flux of a given contaminant over a specified period of time is calculated as the product of the average concentration of the leachate and the amount of leachate produced during that period divided by the length of the period.

When using leaching data as input to transport and behaviour models, it is often convenient to be able to quantify the leaching process in terms of simple mathematical formulas. As discussed elsewhere in these proceedings [13] more or less sophisticated leaching models may be used. The leaching of several (but not all) inorganic contaminants may be described as resulting in an initial or early peak concentration of the contaminant in the leachate followed by an exponential decrease of the concentration with time (or L/S). If it is assumed that a continuously stirred tank reactor (CSTR) model (see [13]) can be used to interpret the results of a column leaching test on the granular waste material, the leaching of several components may be expressed by a simple decay function:

$$C = C_0 * e^{-(L/S)\kappa}$$

where C is the concentration of the contaminant in the leachate as a function of L/S (mg/l), C_0 is the initial peak concentration of the contaminant in the leachate (mg/l), L/S is the liquid to solid ratio corresponding to the concentration C (l/kg) and where κ is a kinetic constant describing the rate of decrease of the concentration as a function of L/S for a given material and a given component (kg/l). κ values may be estimated from column, lysimeter or serial batch leaching data (see [13] in these proceedings).

By integrating the above expression, the amount of contaminant, E (in mg/kg), released over the period of time it takes for L/S to increase from 0 l/kg to the value corresponding to C, can be calculated:

$$E = (C_0/\kappa)(1 - e^{-(L/S)\kappa})$$

The relationship between time and L/S for a percolation flow situation can be derived from the scenario descriptions in step 3 and step 4:

$$t = (L/S) * d * H/I$$

where t is the time since the application started producing leachate, L is the total volume of leachate or percolate produced at time t, S is the total dry mass of waste in the application or layer in question, d is the average dry bulk density of the waste material in the application, H is the average height of the application or the layer in question and I is the net rate of infiltration of precipitation percolating through the application.

For a 0.50 m thick sub-base layer of MSWI bottom ash, L/S = 1 l/kg will correspond to a period of approximately 15 years for a rate of infiltration corresponding 50 mm/year and 2.5 years, if the rate of infiltration is 300 mm/year (assuming a uniformly distributed percolation through the material, which is not likely to occur in practice). For a 5 m thick application of MSWI bottom ash, the corresponding periods needed to reach L/S = 1 l/kg are 150 years and 25 years, respectively.

The flux of contaminants from the base of a granular waste application percolated by infiltrated precipitation is then described as a function of time by substituting the L/S with time in the above expression for C and combining (multiplying) that with the estimated rate of flow of water through the waste material.

Predictions of leaching properties of mineral waste materials over longer time periods may be influenced by mineral changes as well as external factors and should be supported by hydrogeochemical modelling and information on the influence of pH and redox potential on leaching (see e.g. [13]).

Step 6: Description and modelling of the migration of the contaminants from the application to the POC(s)

On the way from the base of the application to the groundwater and the POC(s), the contaminants leached from the inert waste are first transported vertically through the unsaturated zone below the below the application to the groundwater. They are subsequently transported laterally with the groundwater to the POC(s). Various attenuation processes such as dispersion/dilution and interaction with soil/ groundwater (only sorption considered) influence the transport velocity and distribution of the contaminants in the aquifer. The transport behaviour of different contaminants varies widely and is also dependent on the properties of the aquifer. Some contaminants (e.g. chloride) are very mobile and only affected by dilution/dispersion, whereas others (e.g. lead) are almost immobile, even over longer periods of time. These differences in behaviour are reflected by the resulting concentration profiles as a function of time at the POC(s).

For mobile constituents, a direct relationship between peak concentration (mg/l) in a leaching test and the maximum concentration in the groundwater at the POC(s) can be found. This is the case both for locations near and far away from the landfill and reflects the degree of dilution/dispersion in the system. For retarded constituents only POC(s) fairly close to the application are relevant, and the peak concentration in the groundwater near the application generally shows a less straightforward relationship to the peak concentration in the leaching test than the for the mobile constituents. The retention mechanism tends to smooth out the groundwater quality peaks, and peak concentrations in the leaching test may not necessarily appear at the lowest L/S in the test for all contaminants.

Included in this step is also the selection, set-up and coupling of mathematical models describing the contaminant migration, first from the base of the waste application to the groundwater table (unsaturated zone model), then from the aquifer below the application to the point of compliance (saturated zone model). The unsaturated and saturated zone models may be discrete, but coupled, or they may be built into one package (see [11, 13]).

Most state-of-the-art groundwater transport models are based on the same fundamental groundwater transport equations and are expected to give similar results for the same input. This has been shown for a selection of models [11]. The models may, however, differ widely in focus and degree of detail (source description, inclusion/exclusion of the unsaturated zone, inclusion of attenuation processes, groundwater hydrology, general infiltration etc.) as well as in type and solution techniques (1, 2 or 3 dimensional, numeric/analytical, stochastic/ deterministic). The choice of model should depend on the specific objectives of the modelling and a balance between the degree of sophistication of the model and the available input.

The consideration of contaminant/subsoil interaction is strongly recommended, e.g. by inclusion of simple reversible sorption processes and assuming they may be described by linear sorption isotherms (expressed in terms of K_D values for each contaminant), both in the unsaturated and the saturated zones. Linear adsorption is included in most up-to-date groundwater transport models. The K_D values used may be based on literature or, if the models are used for site-specific risk assessment, on specific knowledge of the subsoil in question.

Step 7: Performance of "forward" modelling to determine attenuation factors

In this step, the models that were chosen and adjusted to the situation are run with a variable source input as described in step 5. The sensitivity of the model to variations of important input parameters such as rate of infiltration of precipitation, groundwater flow velocity, thickness of the unsaturated zone, K_D values κ values, etc. should be tested. For a given contaminant, the result of the model calculations may be the concentration of that contaminant as a function of time in the groundwater at the chosen POC. The modelling is continued until a peak has occurred. The ratio, $C_P/C_0 = f$, between the peak concentration of the contaminant at the POC and the peak concentration, C_0, of the source term, may be calculated (= f, where the attenuation factor, AF = 1/f). If the models are run in order to assess the risk in a specific case, site-specific scenario descriptions should be used, and the concentration profile as a function of time at the POC is a measure of the impact and risk in that particular case.

Step 8: Application of the results to criteria setting ("backwards" calculation)

Once the attenuation factors for each contaminant of interest have been determined, the next step in the development of leaching criteria is relatively simple. Criteria, C_{CRIT}, are set for the maximum (peak) values of the concentrations of relevant contaminants as described in step 2. The peak value, C_0 obtained in the test and used in the description of the source term in step 4 is then calculated from the expressions for the attenuation factors determined in step 7: $C_0 = C_P/AF = C_{CRIT}/AF$.

Step 9: Transformation of the source term criteria to limit values at different L/S values

When C0 corresponding to the groundwater criteria, CCRIT, has been determined for a given contaminant, the expressions shown in step 5 for C and E can be used to calculate the corresponding limit values for the result of a batch or column leaching test expressed in terms of concentration of the contaminant in the eluate (CLIMIT, e.g. in mg/l) or amount leached at a certain L/S value (ELIMIT, e.g. in mg/kg). While the value of κ is contaminant (and to some extent material) specific, L/S may be varied in the expressions for CLIMIT and ELIMIT, thus allowing some freedom of choice of leaching test. This is an advantage, because some Member States prefer using regulatory batch leaching tests performed at L/S = 2 l/kg (e.g. EN 12457-1), whereas other Member states prefer regulatory batch leaching tests performed at L/S = 10 l/kg (e.g. EN 12457-2). In this way each EU Member State can prescribe the use of its preferred leaching test at national level and still maintain the same level of environmental protection. The calculated limit values corresponding to each leaching test will be different (depending on L/S) but the resulting maximum level of concentration at the POC will be the same. Such a solution was found when setting the leaching limit values for acceptance of waste at certain types of landfills regulated by the EU landfill Directive and the associated Council Decision on acceptance of waste at landfills [10]. If the peak concentration in the eluate, C0, itself is used as a limit value for leaching, batch leaching tests cannot be used, because for practical reasons they cannot be performed at L/S values much lower than 2 l/kg. The peak concentration generally occurs at much lower L/S values, and C0 may sometimes refer to the concentration of the first fraction of eluate from a column leaching, e.g. representing L/S = 0.0 - 0.1 l/kg.

Test Methods

As already mentioned, it is very important that the mechanisms controlling the release of contaminants from the waste materials are identified correctly and that the test methods prescribed or used are consistent with this information. This means that for granular materials in a percolation scenario, a column leaching test performed under conditions approaching local equilibrium would generally be appropriate when testing at basic characterisation level in the hierarchic system prescribed by CEN/TC 292 and incorporated into the Landfill Directive (Annex II in [9]). Once the leaching behaviour of a material has been established at basic characterisation level, a simpler test may be used for quality control purposes at compliance level, provided the relationship between the results of the two types of tests is sufficiently well established. The column test prEN 14405 provides a detailed description of the leaching of inorganic contaminants from granular waste materials as a function of L/S with 7 points on the leaching curve from L/S = 0.1 l/kg to 10 l/kg. The corresponding compliance test is EN 12457, part 1, 2 and 3, which provides 1 or 2 points on the same leaching curve at L/S = 2 l/kg (part 1), L/S = 10 l/kg (part 2) and L/S = 2 and 10 l/kg (part 3). If the waste material is not granular but monolithic or if it is placed in non-percolation hydraulic scenario, other test methods based on transport by diffusion rather than equilibrium/convection must be applied. This would change the source term and hence the relationships used to transform groundwater quality criteria into limit values for a leaching test by a step-wise scenario-based procedure as the one presented above. Application of a leaching method intended for granular waste materials to monolithic waste or to insufficiently crushed coarse granular waste may lead to meaningless results, if the surface related release by diffusion is not accounted for.

Validation of Risk Assessment Modelling

As stated in ENV 12290 and shown in figure 1, it is important to validate models that are used in the application and interpretation of leaching test results. This is particularly true for impact and risk assessments based on a multitude of simplifications such as the step-wise procedure described above. Both large scale pilot studies and field observations should be exploited for this purpose.

DHI is currently involved in a study aiming at the validation of the impact and risk assessments, upon which the Danish regulation of utilisation of MSWI BA is based [16]. For this purposes a large-scale demonstration site has been established. The demonstration project consists of 6 separate sites, 4 with a surface area of approximately 100 m^2, 2 with a surface area of approximately 200 m^2. All sites contain a 50 to 70 cm sub-base of MSWI bottom ash, compacted in 3 layers as normally done in road construction. All units except one are equipped with LDPE bottom liners and drainage layers as well as pumps and wells for collection of percolating leachate. Bottom ash from 3 different MSW incinerators have been used. Three of the sites have only been covered with gravel, which is intended to optimise the infiltration of precipitation and the production of leachate. The objective here is to study the contamination source as a function of time and L/S. Two other sites have top covers of flagstones and asphalt, respectively. In these cases the major objective is to study the infiltration reduction effectiveness of the cover, including various edge effects. The last site has been constructed without a bottom liner and with only a gravel cover, allowing the leachate formed to leak into the secondary aquifer. Groundwater monitoring wells have been placed both upstream and downstream of this site with the objective to study the migration of the leachate. The MSWI bottom ashes used in the sites are being characterised extensively with respect to composition and leaching behaviour in the laboratory.

Preliminary water balance results indicate that edge effects are important and must be taken into account when the leaching data are interpreted. The laboratory characterisation of the bottom ashes used in the sites, including their leaching characteristics, is still in progress. The data collected from the site and the results of the laboratory work will be used in models to establish realistic descriptions of the release and migration of contaminants for MSWI BA used as sub-base in roads and to validate the models used by the Danish Environmental Protection Agency.

Part 2: Aspects for selection of models and examples from modelling work

Approach in modelling

Once the source term, i.e. the flux of contaminants as a function of time, has been determined (model 1), the migration of the contaminants from the border (at the base) of the landfill/the utilisation application through the unsaturated zone into the groundwater must be described (model 2), and finally, the transport of the contaminants in the groundwater through the aquifer to a ”point of compliance” (POC), is described by a third model (model 3). The POC in figure 5 may be a point located a certain distance downstream of the landfill at which the impact is determined or where specific groundwater quality criteria must be met. If the target is a surface water body, a fourth model describing the onward dilution and transport of the contaminants in a given surface water body may be added. As described below, the models used may be more or less sophisticated and they may or may not include various phenomena, which influence the transport of contaminants. The models may be set up to describe relatively general sceanarios and situations as it was done in the setting of acceptance criteria for landfilling of waste in the EU (Hjelmar et al., 2001). In this case, general or average/critical input parameters are used. However, the same principles and the same models may be used for site-specific impact assessments or even development of site-specific waste acceptance criteria, if site-specific input parameters and data are used

A general description of the entire procedure of setting criteria for the acceptance of waste at a landfill or utilising waste materials for construction purposes is described in Appendix 4. Here also the parameter values used in the modelling of the contaminant transport in the saturated zone in the calculations of the EU criteria for acceptance of waste at landfills are also shown.

Groundwater flow and transport modelling

Percolated leachate will be transported both in the unsaturated zone below the landfill and in the groundwater depending on the composition of the leachate and its geochemical and chemical properties. It is important to describe the hydrology of the sub-surface system correctly in order to simulate the transport and attenuation of solutes correctly. Flow conditions in the subsurface system are often very complex and a detailed geological and hydrogeological understanding is required to be able to simulate subsurface flow. Driving variables in this system are meteorological conditions in terms of precipitation, evaporation and evapotranspiration – the latter depending on soil and land use conditions as well as specific crop characteristics. Groundwater recharge is as such highly variable both in time and space and need careful determination in order to estimate unsaturated and groundwater flow. Groundwater flow is governed by hydraulic characteristics of the geological formations as well as “boundary conditions” in terms of rivers, lakes and other recipients. Stresses to the groundwater in terms of e.g. groundwater abstraction and forced infiltration may also affect groundwater flow patterns.

Flow conditions are as such often highly dynamic and with large spatial variations. Therefore, it is recommended that physically based models are applied with as few lumped parameters as possible. This means that measurements can be utilised directly as data source for the model. Anyway, a careful calibration of the flow model against measurements of groundwater head, soil water content, fluxes and water levels in recipients is required before any transport modelling is initiated.

Transport of solutes in the subsurface system depends on the flow conditions as described above. Furthermore, the effective porosity, dispersion conditions and the chemical and geochemical characteristics of the soil and solutes will affect the movement of solutes. Such information is often difficult to obtain and calibration against measurements of solute concentrations in groundwater and recipients, water age etc. will improve the model reliability.

Application of groundwater flow and transport models is, however, the only way to analyse flow and transport phenomena and remediation alternatives towards groundwater pollution. Even though many parameters are associated with large uncertainties analyses of alternative transport scenarios may give many answers to “what if” questions.

Theory

A large number of input data is required to develop a reliable subsurface flow, transport and reactive model to describe the transport and fate of landfill leachate.

Flow conditions in the unsaturated zone are traditionally described by Richard´s equation which is examined in many textbooks, see e.g. Maidment (1992). It describes the water flow, soil water content and hydraulic conductivity distribution as a function of depth and time. Boundary conditions are typically recharge and depth to the groundwater table and it is often assumed that the flow in the unsaturated zone is vertical thus a one-dimensional modelling approach can be valid.

Groundwater flow is a three-dimensional phenomena governed by Darcy´s law combined with a mass balance condition. The governing equation for three-dimensional dynamic groundwater flow may be described as:

$$\frac{\partial}{\partial x}(K_{xx}\frac{\partial h}{\partial x})+\frac{\partial}{\partial y}(K_{yy}\frac{\partial h}{\partial y})+\frac{\partial}{\partial z}(K_{zz}\frac{\partial h}{\partial z})-Q=Ss\frac{\partial h}{\partial t}$$

Where K is hydraulic conductivity [L/T], h is potential head [L], Q is boundary flux [L^3/T/L^2] (in the case of pumping), S_s is the specific storage coefficient (L^2/L^3) and x, y, z are spatial co-ordinates [L] and t is time [T]. Groundwater flow models often solve this equation either in a finite-difference or a finite-element grid system. The solution to this equation for potential head and the flow velocities or fluxes required for solute transport simulations are determined from Darcy´s law:

$$Q = \Delta hC$$

Where Δh is the head difference [L[, C is the conductance [L/T/M] and Q is the flux [L^3/T/L^2].

Sub-surface flow models require collection and processing of the following temporal and spatial varying input data:

- Soil properties in terms of retention curve and hydraulic conductivity function
- Hydrogeological conditions i.e. hydraulic conductivity matrix and porosity
- Precipitation/evaporation and evapotranspiration
- Boundary conditions in terms of rivers, lakes and other recipients
- Stresses to the system in terms of groundwater pumping or forced injection
- Calibration data in terms of measured potential head, fluxes in recipients, soil water profiles

Reactive solute transport is also a highly dynamic and spatial varying phenomena governed by the advection-dispersion equation, which may be formulated as:

$$\frac{\partial c}{\partial t}=-\frac{\partial}{\partial x_i}(cv_i)+\frac{\partial}{\partial x_i}\left(D_{ij}\frac{\partial c}{\partial x_j}\right)+R_c \quad i,j=1,2,3$$

Where c is concentration of solute [M/L^3], v is pore water velocity [L/S], D is dispersion tensor [L^2/T], x is spatial co-ordinates [L], t is time [T] and Rc is a source/sink term which may also stem from reactions [M/L^3/T]. The equation describing transport in the unsaturated zone is in principle the same but often simplified to one dimension. Solute transport models require estimation and/or collection and processing of the following temporal and spatially varying input data:

- dispersion characteristics of solutes and soils
- degradation of solutes
- geochemical characteristics
- geochemical reactions
- calibration data in terms of measured concentrations

Often geochemical reactions are described by macro parameters such as retardation coefficients and degradation (decay) constants. Such parameter values are often found in the literature for specific solutes under different soil conditions, but may need to be subjected for calibration if concentration measurements have been carried out. Often K_d and log K_{ow} may be used to determine retardation factors.

Available models

A large variety of unsaturated zone and groundwater flow, transport and reactive transport models are available on the marked. In table 1 below some of the most commonly used models are listed with an explanation of their capabilities.

Table 1. Some commonly used groundwater transport models.

Name	Description	Reference
WHI Unsat Suite	Combines four models – SESOIL, VLEACH, VS2DT og PESTAN – for the analyses of one-dimensional flow and transport through the unsaturated zone.	Waterloo Hydrogeologic: Groundwater & Environmental Catalog, 2004.
FEFLOW	Finite-element model for the description of flow and solute transport including reactive transport in groundwater and the unsaturated zone. Furthermore, variably density as well as heat transport may be simulated.	Waterloo Hydrogeologic: Groundwater & Environmental Catalog, 2004.
Visual MODFLOW Pro	Models (MODFLOW; MODPATH, RT3D, RT3DMS) for the 3D simulation of groundwater flow and solute transport including multi-species reactive transport and particel tracking.	Waterloo Hydrogeologic: Groundwater & Environmental Catalog, 2004.
MOFDLOW-SURFACT	Finite-difference model for the simulation of coupled unsatuted/saturated flow and solute transport including multi-species reactive transport in different phases and two domains.	Waterloo Hydrogeologic: Groundwater & Environmental Catalog, 2004.
FLOWPATH II	Two-dimensional groundwater model for the simulation of flow and particel tracking for the analyses of solute transport, flow paths and capture zones..	Waterloo Hydrogeologic: Groundwater & Environmental Catalog, 2004.
FRACTRAN	To-dimensional, finite-element model for the simulation of steady groundwater flow and solute transport in fractured media.	Waterloo Hydrogeologic: Groundwater & Environmental Catalog, 2004.
FRAC3DVS	Finite-element model for the simulation of unsaturated and groundwater flow and transport in porous and fractured media.	Waterloo Hydrogeologic: Groundwater & Environmental Catalog, 2004.
MIKE SHE	Three-dimensional finite-difference model for the simulation of unsaturated and saturated flow and transport including multi-species, reactive transport in two domains.	www.dhi.dk
Risc WorkBench	Software package for the simulation of transport and fate of solutes including human risk assessment. Follows US EPA standards.	Waterloo Hydrogeologic: Groundwater & Environmental Catalog, 2004.

When choosing a model to solve a particular problem, several aspects may be considered. The model should, of course, be capable of solving the the problem at hand. Several models are, for example, not able to use a variable input source function. It is, on the other hand, not always necessary to use a very sophisticated model to solve a relatively simple problem. The model should be chosen to fit the problem and also to fit the quality and availability of data and also the vulnerability of the situation. If the risk or its consequences is low, a sophisticated model is probably unwarranted.

The more sophisticated a model gets, the more input data are usually required. If limited or no specific data are available, then the usefulness of sophisticated model functions requiring such data is limited. It should also be considered that very sophisticated, multi-dimensional numerical models generally gets slower with increasing complexity. In some simple cases with symmetric characteristics, 2-D models may be applied to solve 3-D problems. However, if sufficient data of good quality are available, a more sophisticated model will generally be able to take more aspects of release and transport into account and hence produce a better result.

Table 2 provides an overview of the models that were tested (and found useful) by the Technical Adaptation Committee (TAC) in setting criteria for the acceptance of waste at landfills in accordance with the EU Landfill Directive (Hjelmar et al., 2001). The actual calculations were performed using CXTFIT/ECOSAT and HYDRUS 2D for the unsaturated zone and MODFLOW and MT3D for the saturated zone.

Table 2. Overview of transport models applied to the scenarios.

Model	Main features	Waste release	Unsaturated zone	Soil interaction	Groundwater hydrology	Type of model	
MISP (BRGM)	General groundwater transport	Decay function	+	Kd	Column-aquifer	Analytical	Semi 3 D
ECOSAT (ECN)	Detailed soil chemistry	Various options	- +chemistry	Kd, extended chemistry	Column	Numeric	1D
LANDSIM (Golder Ass.)	Specific for landfill	Simple decay or constant source	+	Kd	Column-aquifer	Analytical, stochastic	2D
GW Vistas-2 (DHI)	Detailed hydrology	Decay function	+ (+ CXTFIT 2.0)	Kd	Column-aquifer	Numeric, stochastic	3D

Typical input data

Table 3 shows the scenarios used to describe the landfills (and the unsaturated zone, since in the calculations carried out with ECOSAT and GW Vistas (but not in those using LANDSIM)) it was identical with the clay liner at the bottom of the landfill) in the calculations of the EU criteria for acceptance of waste at landfills.

Table 4 shows the parameter values used in the modelling of the contaminant transport in the saturated zone in the calculations of the EU criteria for acceptance of waste at landfills.

Table 3. Description of landfill scenario conditions and assumptions used in the TAC calculations (Hjelmar and Holm, 2004).

Parameter	Unit	Inert waste landfill	Non-hazardous waste landfill	Hazardous waste landfill
Height of the landfill	m	20	20	20
Length of the landfill	m	150	200	200
Width of the landfill	m	150	200	200
Surface area	m^2	22500	40000	40000
Volume	m^3	450000	800000	800000
Porosity of the waste	-	0.3	0.3	0.3
Dry bulk density of the waste	t/m^3	1.5	1.5	1.5
Dry weight of the waste	t	675000	1200000	1200000
Permeability of the waste	m/s	1×10^{-5}	1×10^{-5}	1×10^{-5}
Hydraulic conductivity of top cover	mm/a	> 300	variable: (0-200) composite*	variable: (0-200) composite*
Type of bottom liner	m	none	1	5
Thickness of bottom liner (clay)	m	-	1×10^{-9}	1×10^{-9}
Permeability of clay bottom liner	m/s	-		

*: composite bottom liner (artificial liner + clay liner)

Table 4. Parameter values used in the model calculations of transport in the saturated zone in the calculations performed by the TAC (Hjelmar and Holm, 2004).

Parameter	Unit	Used by TAC
Width of catchment	m	500
Length of catchment	m	600
Distance from water divide to beginning of landfill	m	100
Distance to POC	m	20 and 200
Net rate of infiltration	mm/year	300
Thickness of aquifer	m	Approx. 6
Upper boundary	-	Closed
Fixed hydraulic head at downstream boundary	m	Approx. 4.1
Horizontal hydraulic conductivity Kx = Ky	m/s	1.4×10^{-4}
Vertical hydraulic conductivity Kz	m/s	1.4×10^{-4}
Effective porosity	-	0.3
Longitudinal dispersivity	m	20
Transversal dispersivity	m	4
Vertical dispersivity	m	2
Cell size	m	10
Number of calculation layers	-	6

Examples of modelling results

Table 5 shows some examples of the use of three models in series (source, unsaturated zone, saturated zone) as described earlier. The calculations have been carried out for three different scenarios and the results are expressed in terms of the maximum allowable leached amonts in a batch leaching test carried out L/S = 10 l/kg that will ensure that the peak concentration occurring at the POC will not exceed the groundwater quality criteria (drinking water standard or groundwater protection standard).

In Case A, an unsaturated zone thickness of 5 m and a groundwater flow of 20 m/year are assumed. Drinking water criteria are used as the reference quality criteria at the POC. In Case B the assumed thickness of the unsaturated zone is reduced to 1m and the groundwater flow in the aquifer is reduced to 4 m/year. The reference quality criteria are unchanged. Case C is similar to Case B, except that the drinking water standards used as reference quality criteria at the POC are changed to more restrictive (lower) groundwater quality protection standards.

Table 5. Examples of some of the results of the modelling in support of the setting of EU limit values for acceptance of waste at landfills. The results shown are limit values corresponding to a leaching test carried out at L/S = 10 l/kg.

		ECN-DHI-modelling	
Reference value for acceptance	**drinking water standard**	**drinking water standard**	**protection of groundwater quality**
Groundwater flow	**20m/yea**	**4m/year**	**4m/year**
Unsaturated zone	**5 m r**	**1 m**	**1 m**
Infiltration	**300 mm/y**	**300 mm/y**	**300 mm/y**
	Case A	**Case B**	**Case C**
Sb, mg/kg	0,08	0,05	0,01
As, mg/kg	0,28	0,14	0,28
Ba, mg/kg	8,76	5,6	0,8
Cd, mg/kg	0,07	0,02	0,002
Cr, mg/kg	3,61	0,7	0,36
Cu, mg/kg	32,8	12	0,31
Hg, mg/kg	0,02	0,01	0,002
Pb, mg/kg	0,41	0,1	0,28
Mo, mg/kg	0,98/1,66*	0,4/0,75	0,10/0,21
Ni, mg/kg	0,83	0,2	0,09
Zn, mg/kg	81,8	23	2,30
Se, mg/kg	0,09/0,17	0,04/0,09	0,01/0,01
F^-, mg/kg	15,3	9	4,4
SO_4^{2-}, mg/kg	1639/2971	1110/2320	555/1160
Cl^-, mg/kg	996/1805	670/1390	400/836

*POC at 20 m/200 m distance from the edge of landfill

It can be seen that with a few exceptions, the resulting limit values get more stringent (lower) in the order Case A > Case B > Case C. The main reasons that Case B criteria come out more restrictive than Case A criteria are that the reduced groundwater flow velocity and to a lesser degree the reduced thickness of the unsaturated zone (reduced retention capacity) causes a reduction in the dilution and hence an increase in the impact at the POC. The reason Case C in general generates more restrictive criteria than Case B is that groundwater quality criteria are meant to protect the entire ecosystem an therefore in most cases are substantially lower than drinking water criteria.

Uncertainties of evaluation

Numerous uncertainties are involved in the use of scenarios, site descriptions, coupled models, and assumed or averaged parameters and data. It is important to evaluate this uncertainty and this will be pursued further in the project. Experience has shown that the assumptions made concerning the water balance (resulting from a particular design and operation of the landfill) as well as the leaching behaviour of the waste materials have a particularly dominant influence on the results. Several other factors are also important, including the dilution and retention capacity of the aquifer. Some sensitivity analyses are available and will be discussed below.

Ref.

- Hjelmar, O., H.A. van der Sloot, D. Guyonnet, R.P.J.J. Rietra, A. Brun & D. Hall (2001): Development of acceptance criteria for landfilling of waste: An approach based on impact modelling and scenario calculations. In: T.H. Christensen, R. Cossu and R. Stegmann (eds.): Sardinia 2001, Proceedings of the Eigth International Waste Management and Landfill Symposium, S. Margharita di Pula, Cagliari, CISA , Vol. III, pp. 712-721, CISA, Cagliari.
- Hjelmar, O. and Holm, J. (2004): Danish implementation of the waste acceptance criteria for landfilling. In the Proceedings of Waste 2004, Stratford-upon-Avon, Warwickshire, England, 28 – 30 September 2004, The Waste Conference Ltd, Coventry, UK.

Appendix 7: Checklist for modelling

When preparing a modelling exercise to study the feasibility of applying the factor 3 rule, the following checklist may be of some use. It is assumed that a problem waste and a relevant landfill have been identified.

- Identify the problem, i.e. determine which for which components the WAC are exceeded.
- If the components identified under 1) are not TOC/DOC, Cd or Hg, and the leaching results for the other regulated components do not exceed 3 x WAC, use of the factor 3 rule may be considered.
- Obtain the necessary additional waste-specific and site-specific information.
- Describe the landfill scenario in detail.
- Describe the environmental scenario in detail, including the unsaturated zone, the aquifer, any nearby surface water bodies, the appropriate POC and WQC, etc.
- Obtain a suitable (set of) transport model(s) and add a source term model based on (column) leaching data for the components in question and infiltration/leakage data for the landfill to describe the pathway of the contaminants from the landfill to the POC.
- Run the model(s) in a forward mode and obtain baseline attenuation factors for the components in question. If actual waste-specific leaching data are used for the source term, it should be checked whether the peak values at the POC exceed the WQCs. If they are not exceeded (with a good safety margin), the exercise could possible be discontinued since compliance with the factor 3 rule is indicated.
- If the WQCs in 7) are exceeded moderately or if the margin of compliance is very narrow, further modelling could be considered. The attenuation factors should be calculated.
- If it is realistic to change some of the design or operation conditions of the landfill (in particular conditions that would affect the water balance and reduce the leakage rate), the effect of this could be evaluated as described in section 5.4, i.e. the model is run again with the changes incorporated. If this indicates that the changes will

provide sufficient increases in the attenuation factors, then the application of the factor 3 rule may be recommended.

- If the model calculations performed under 9) do not lead to sufficient increases in the attenuation factors, then application of the factor 3 rule cannot be recommended for that waste material at that landfill.