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Summary

A Nordic study has been carried out to develop a methodology for the setting of criteria for acceptance of monolithic waste at landfills in accordance with the requirements in the Council Decision 2003/33/EC, to recommend appropriate methods to test monolithic waste and to propose associated criteria/limit values for acceptance of monolithic waste at landfills. The main conclusions of the study may be summarised as follows:

“Monolithic waste” is not clearly defined in the Landfill Directive and the Council Decision 2003/33/EC. Most of the existing definitions of “monolithic” waste refer to certain minimum dimensions and physical and mechanical properties of a test body that ensure its integrity over a certain period of time (usually the duration of a so-called tank leaching test).

Generally, “monolithic waste” has been perceived as waste occurring in large blocks or large coherent masses of concrete-like material of substantial strength and very low permeability. It has further been expected that the release of contaminants from monolithic waste material placed in a landfill could be tested and described by means of a tank leaching test performed on test bodies of the monolithic material in question under dynamic (diffusion or release rate controlled) conditions. The results of the study have, however, indicated that waste landfilled as being “monolithic” generally appears to be less “hard” and coherent than originally assumed, and that the composition of the leachate at the bottom of a typical monolithic landfill for most contaminants is likely to be controlled by equilibrium rather than release rate.

The study has further shown that the release behaviour of “monolithic” waste depends not only upon the characteristics of the waste itself (the rate of release and the solubility of the contaminants) but also on the landfill scenario, in particular the geometric surface area per unit volume of the waste material in the landfill, the height of the landfill and rate of infiltration and percolation of precipitation through the landfill. A simplified model has been set up to test whether or not the leachate composition at the base of a landfill containing a given “monolithic” waste material in a given scenario is likely to be controlled by equilibrium or by the rate of release of contaminants from the solid phase. Application of this model on three types of solidified, monolithic waste, for which leaching data were available, indicated that for nearly all leachate components measured, the concentration in the leachate at the base of the landfill would appear to be controlled by equilibrium over a wide range of variation of the above mentioned parameters. This means that the physical structure

of the monolithic waste has little effect on the composition of the leachate under the conditions studied and that the leachate composition basically is the same as that which would be expected for the same material in a granular form.

The methodology used by the TAC to develop waste acceptance criteria (WAC) for landfilling of granular waste materials is presented and discussed in some detail in the report. In order to ensure the same level of environmental protection for landfilling of monolithic waste as for granular waste, the same methodology should be used, only with the source model adjusted (if needed) to fit monolithic waste rather than granular waste. Various model inputs and parameters may be adjusted to fit national conditions, if they differ significantly from those used by the TAC. The same methodology may also be used with local and more specific input data and parameters for site-specific impact assessments.

Based on the above, it is apparent that if the same basic landfill scenarios as those applied by the TAC are used for landfills for hazardous and non-hazardous monolithic waste (or non-hazardous waste landfills receiving stable, non-reactive hazardous monolithic waste), the appropriate level of protection would be achieved if the monolithic waste is crushed to < 4 mm and tested the same way as required for granular waste and the results of the tests are compared to the same WAC as those for granular waste in 2003/33/EC. A waste owner testing his waste would then benefit from the chemical effect of a stabilisation but not from the physical effect, since the latter appears to have little or no effect on the composition of the leachate at the bottom of the landfill under normal conditions. If the water balance is different from that of the TAC modelling, the models could be run again to find adjusted values (for the purpose of illustration, this has actually been done for water balance conditions corresponding to those required in Sweden). Similarly, values could need to be adjusted, possibly by re-running the models, if the downstream requirements are changed (the groundwater quality criteria may be changed or the modelling area may be expanded to include surface water bodies).

It is interesting to notice that during the negotiations in the TAC, the acceptance criteria that actually resulted from the modelling were increased for all parameters (except Cr) to obtain the values listed in 2003/33/EC. The factors by which they were increased range from 1.1 to 17 for non-hazardous waste landfills and from 1.6 to 24 for hazardous waste landfills.

WAC for landfilling of monolithic waste have been set or proposed in the United Kingdom, the Netherlands and France, based on other criteria and test methods (mostly on tank leaching tests). Sufficient test results were available for one monolithic waste material to compare the performance in terms of compliance/non-compliance of the material in the three systems (for hazardous waste landfills) as well as in the system

existing for granular waste (based on testing of crushed monolithic material). The comparison shows that the proposed UK criteria are most restrictive whereas the Dutch and the French criteria are more lenient. For the waste tested, the performance was practically the same in the Dutch and French system as well as in the “granular” system, where the material was crushed prior to testing and evaluated against the criteria for granular waste.

Although the model considerations used are simple and the results should be confirmed by more sophisticated models and by field observations, the following recommendations may be presented:

- In view of the limited information and experience available and the ongoing efforts to develop better test methods and models that better describe the release of components in a monolithic landfill, it is recommended to keep an “open mind”, leaving room for implementation of new developments and improvements.
- At this stage, it is recommended not to distinguish between “granular” and “monolithic” waste materials as such in relation to the setting of waste acceptance criteria landfilling and testing of compliance with these criteria as prescribed in Council Decision 2003/33/EC.
- It is recommended to crush “monolithic” waste (or waste with particle size above 4 to 10 mm) to a particle size < 4 mm and test the leaching properties using the same methods as those prescribed for granular waste (in particular EN 12457-part 1-3 and CEN/TS 14405). CEN/TS 14405, which is a column test, includes the option of testing a material with a particle size < 10 mm provided a large column diatemer (10 cm) is used.
- If the TAC scenarios are used in the model calculations without changes, the same WAC as those listed in Council Decision 2003/33/EC for granular waste should apply. If changes are made in input parameters to the models or the scenarios are changes (e.g. in terms of water balances for the landfills or downstream water quality targets and target values), the models must be re-run to obtain new criteria.

1. Introduction

In the Council Decision of 19 December 2002 establishing criteria and procedures for the acceptance of waste at landfills pursuant to Article 16 and Annex II of Directive 1999/31/EC on the landfill of waste (2003/33/EC), criteria and limit values are set for the acceptance of granular waste materials at certain classes and sub-categories of waste. Specific criteria have not been set for monolithic waste at EU level, but it is stated that Member States must set criteria for monolithic waste. It is further stated that such criteria must provide the same level of environmental protection as that provided by the corresponding limit values for acceptance of granular waste at a given class or sub-category of landfill. In the decision document, development of national criteria for acceptance of monolithic waste is required for landfills for hazardous waste and landfills for non-hazardous waste receiving stable, non-reactive hazardous waste.

To achieve this, the criteria for acceptance of monolithic waste must be developed using the same scenario framework as that used for the development of criteria for granular waste. However, the part of the scenario describing the landfill itself and the source term must be based on release of contaminants from monolithic waste and may differ from that for granular waste.

The Nordic Council of Ministers has contracted with DHI – Water & Environment (DHI) in co-operation with the Swedish Geotechnical Institute (SGI) and the Technical Research Centre of Finland (VTT) to carry out a study to support the efforts of the landfill authorities in the Nordic countries in developing criteria for landfilling of monolithic waste. The objectives of the study are:

- To develop a methodology for the setting of criteria for acceptance of monolithic waste at landfills in accordance with the requirements specified in the Council Decision of 19 December 2002 establishing criteria and procedures for the acceptance of waste at landfills pursuant to Article 16 and Annex II of Directive 1999/31/EC;
- To recommend appropriate methods to test monolithic waste and to propose associated criteria/limit values for acceptance of monolithic waste at landfills. The criteria must provide the same level of environmental protection as the criteria for granular waste listed in the Council Decision 2003/33/EC.

A steering group consisting of the members of the Landfill Group under the Nordic Council of Ministers has followed the project. During the

project, major results and conclusions have been made available to the steering group.

This report presents the results of the project. The opinions and views expressed are those of the authors and do not necessarily reflect the views of the national environmental authorities of the Nordic countries.

2. Approach

Although the Council Decision (2003/33/EC) refers to monolithic waste, the term “monolithic” has not been defined in the Council Decision or in the Landfill Directive. As described below, “monolithic” is not a very precisely defined property. Generally, “monolithic waste” has been perceived as waste occurring as large blocks or large coherent masses of concrete-like material of substantial strength and very low permeability (this is consistent with the reference in Article 6 (c) of the Landfill Directive to solidified and vitrified waste). It has implicitly been assumed that the concentration of contaminants in the leachate from “monolithic waste” in a landfill would be controlled by the release rate (often the rate of diffusion), and that a so-called tank leaching test would be the appropriate method for determining the rate of release of contaminants as a function of time (mg/s/m^2 of surface area).

However, in the early stages of this project it became necessary to revise this perception of “monolithic waste” in relation to landfilling and testing. Observations and scenario/model calculations indicated that landfilled waste considered “monolithic” generally appears to be less “hard” and coherent than originally assumed, and that the composition of the leachate at the bottom of a landfill for most contaminants is likely to be controlled by equilibrium rather than release rate. Under those circumstances, an equilibrium-based leaching test such as a column or batch test would be expected to provide a better description of the release of contaminants than a tank leaching test would. As a result of this, the original project plans were somewhat reoriented to focus more strongly on the equilibrium-based situation and to test the validity of the underlying assumptions.

The contents of the report are briefly described below.

Some of the various definitions of “monolithic” and their applicability to landfilling and criteria for landfilling are discussed in **Chapter 3**.

Testing of monolithic waste, both in terms of leaching and in terms of establishing the monolithic character of the waste, is discussed in **Chapter 4**. The availability of standardised test methods and the development in European standardisation of leaching tests for monolithic waste is also discussed. Recommendations are made of leaching tests for characterisation and compliance in relation to landfilling of monolithic waste.

Chapter 5 presents the methodology used by the EU Subcommittee on Landfilling of the Technical Adaptation Committee (TAC) for Waste Related Directives to set criteria for acceptance of granular waste at land-

fills. A similar approach using an appropriate source term should be used to set criteria for monolithic waste in order to ensure the same degree of protection as the acceptance criteria for granular waste.

Chapter 6 provides a brief discussion of the types of “monolithic” landfills that currently exist in the Nordic countries. Some experiences from “monolithic” landfills in other countries in Europe are also reported.

Chapter 7 presents a simplified model, which may provide an indication of whether the composition of the leachate at the base of a “monolithic” landfill is controlled by the rate of release/diffusion out of the solid matrix or whether it is controlled by solubility or (local) equilibrium between the solid phase and the aqueous phase (the leachate). The factors considered in the model are: C_{eq} , the solubility of a given component in a given waste material (low values of C_{eq} favours equilibrium), k , the rate of release of a given component from the surface of a given monolithic material (a high value of k favours equilibrium), A , the geometric surface area per unit volume, which is a measure of the number of cracks in the landfill or the (active) surface of a landfilled material per unit volume (a high value of A favours equilibrium), H , the height of a landfill (a high value of H favours equilibrium) and Inf , the rate of infiltration of precipitation into the landfill (a low value of Inf favours equilibrium). Using experimental leaching data for three different stabilised and solidified waste materials scenario calculations are presented to show under which conditions the composition of the leachate may be assumed to be equilibrium controlled.

Since the scenario calculations indicate that equilibrium control is likely to occur in many “monolithic” landfills, the calculation of the source term under equilibrium conditions (corresponding to the methodology used by the TAC) is described in some detail.

In **Chapter 8**, acceptance criteria for landfilling of monolithic waste corresponding to the equilibrium-based situation are presented, both in terms of values calculated using the same methodology as that used by the TAC in setting criteria for acceptance of granular waste, and in terms of the actual limit values set by the TAC and listed in Council Decision 2003/33/EC. Criteria corresponding to other conditions than those adopted by the TAC are also discussed.

Chapter 9 provides a comparison of the criteria for acceptance of monolithic waste at landfills proposed or adopted in the United Kingdom, the Netherlands and France with each other and with the criteria corresponding to those for granular waste listed in Council Decision 2003/33/EC, after performing a column or batch leaching test on the crushed monolithic material.

The main conclusions and recommendations are summarised in **Chapter 10**.

3. Definition of “monolithic” waste

As already mentioned in Chapter 2, neither the Landfill Directive nor the Council Decision of 19 December 2002 provide any definitions of “monolithic” waste. This is unfortunate in view of the requirement in the Council decision that member states must set national criteria for acceptance of “monolithic” waste at landfills. Generally, “monolithic waste” has been perceived as waste occurring as blocks or large coherent masses of concrete-like material of substantial strength and very low permeability (this is consistent with the reference in Article 6 (c) of the Landfill Directive to solidified and vitrified waste).

It has implicitly been assumed that the concentration of contaminants in leachate from “monolithic waste” in a landfill would be controlled by the rate of release (often the rate of diffusion), and that a so-called tank leaching test would be the appropriate method for determining the rate of release of contaminants as a function of time ($\text{mg}/\text{m}^2/\text{day}$). Most of the existing definitions of “monolithic” materials are related to the test methods (tank leaching tests) and aim to ensure that the release is not affected by the geometry and size of the test body and that the tested material maintains its integrity throughout the test period. Some definitions or requirements also address the intended use of the monolithic material (e.g. compressive strength requirements). Other definitions include permeability requirements, specifying maximum permeabilities that will severely restrict percolation through the material under normal conditions.

Within the context of the work of CEN/TC 292, the definition and requirements of a “monolithic” waste material have been discussed in several papers, e.g. Fällman & Elander (1996), van der Sloot (1996) and Fällman et al. (1996). There is a general agreement that several aspects are important and must be considered/tested to determine the monolithic character of a waste material. These aspects include:

- The release mechanism for contaminants
- The permeability and other intrinsic properties such as pore volume and tortuosity
- The physical durability
- The chemical durability

In relation to testing, typical examples of definitions are:

Monolithic waste: Material that has certain minimum dimensions and physical and mechanical properties that ensure its integrity over the duration of the test (prEN WI292040: 2004)

Monolithic waste: Material which has certain minimum dimensions and physical and mechanical properties that ensure its integrity over a certain period of time (Note: For the purpose of this standard, this period of time is the duration of the compliance test) (CEN/TC 292 WG2, WI292010: 2004).

Test portion of monolithic waste of regular shape: Monolithic waste for which the whole area of the test portion can be calculated on the basis of simple geometric formulas (CEN/TC 292 WG2, WI292010: 2004).

Monolithic test piece: A solid material in the form of a single piece for which specified criteria for dimensions and /or physico-mechanical properties apply (CEN/TC 51 and CEN/TC 104: A study of the characteristic leaching behaviour of hardened concrete for use in the natural environment, Technical report, Third draft, January 1999).

As can be seen, all of these definitions are rather vague and tend to focus on test requirements rather than general requirements. In many tests, the minimum size of the monolithic specimen is a diameter or edge length of 4 cm. Cylinders or cubes with diameters and/or side lengths of 10 cm are often preferred.

The perception of “monolithic” waste as a hard, concrete-like material and the use of the tank leaching test in which the concentration in the aqueous phase is minimised during the test period to assess the rate of release of contaminants were originally developed in the nuclear field to assess a worst case scenario in which vitrified radioactive waste was submerged into water, which was exchanged at a rate that would prevent a substantial build-up of concentration of contaminants in the water phase. This scenario may still have some validity in relation to certain utilisation applications, where a strong brick- or concrete-like material is placed in non-stagnant water, e.g. in a river or in the sea. A tank leaching test, which is designed to maximise the driving force for the release of contaminants from the test body into the water phase, will more or less simulate this scenario. But it is difficult to apply the results of the tank leaching test to the situation in a landfill, where the conditions are completely different, and a build-up of concentrations in the aqueous phase is likely to occur (see Chapter 7). Even though various models are able to describe the release of contaminants from concrete and cement stabilised systems under conditions that include a build-up of concentrations in the aqueous phase, the feasibility of describing the source of contamination or the release of contaminants as a function of time from landfilled “mono-

lithic” waste in a manner that can be used to set general, risk-related waste acceptance criteria and to test compliance with these on the basis of the results of the tank leaching test has not yet been demonstrated satisfactorily.

Since there are indications that many stabilised waste materials that are currently landfilled (see Chapter 6) do not have the hard, concrete-like appearance generally associated with “monolithic” properties, and since the scenario calculations in Chapter 7 indicate that the composition of the leachate at the base of a “monolithic” landfill is likely to be equilibrium or solubility controlled rather than controlled by the rate of release of contaminants from the waste, **it is proposed not to distinguish between “granular” and “monolithic” waste materials as such in relation to the setting of waste acceptance criteria landfilling and testing of compliance with these criteria as prescribed in Council Decision 2003/33/EC.** In the general case, the same criteria (provided the water balance is unchanged) and the same test methods as those applied to granular waste should be required for all waste (implying crushing of waste forms exceeding 4 to 10 mm). However, in order not to block new developments, the national authorities in a country could perhaps choose to refer to the leaching regime in the landfill (equilibrium controlled or release rate controlled), leaving open the opportunity for a landfill owner/operator to perform a risk assessment and prove, on a site-and waste-specific basis, that another test method (e.g. a tank leaching test) referring to other acceptance criteria would be appropriate and would lead to the same level of protection as that represented by the criteria for granular waste listed in the Council Decision 2003/33/EC. This opportunity would, of course, also apply to national authorities in a country in a more general manner.

Should the need arise to test a monolithic waste material in a tank leaching test under dynamic conditions (release rate control), it is proposed that the definition and requirements of “monolithic waste” or “monolithic test piece” of the method or standard in question should apply for the purpose of testing. If the results are applied to landfilling, it is proposed that the leaching scenario and actual requirements on permeability, bearing capacity/compressive strength and durability shall be assessed and evaluated on a waste-and-site-specific basis.

4. Testing of monolithic waste

4.1 CEN/TC 292: Standardisation of leaching tests for monolithic waste

In a mandate from the European Commission (EC, 2002), the European standardisation committee CEN/TC 292 “Characterisation of waste” has been charged with the development of the standard methods needed to carry out the tests required in Council Decision 2003/33/EC. Because of the requirement in the Council Decision that the member states shall set national criteria for landfilling of monolithic waste, the mandate includes an obligation for CEN to produce a dynamic monolithic leaching test for inorganic constituents (a characterisation test) and a compliance leaching test for inorganic monolithic waste of regular shape.

With respect to leaching, R&D and standardisation efforts have focused on the determination of the mechanism of release of contaminants from monolithic materials as described above under conditions aiming at maintaining a maximum driving force. It is usually attempted to keep the concentration of the contaminants in the water in contact with the monolithic specimen as low as possible by frequent replacement with clean water. The most common test method is a so-called tank leaching test, where the monolithic test specimen is suspended in a volume of water, which is replaced with new water at increasing time intervals. The removed water fractions are filtered and analysed for the components of interest. The results are expressed in terms of fluxes of contaminant per unit surface area of the test specimen (e.g. mg/day/m²). The shape (slope) of a log-log graph showing the flux or the accumulated leached amount of a contaminant versus time will indicate whether the release is controlled by surface wash-off/dissolution, diffusion or depletion. In combination with a determination of the amount of a given component available for leaching (obtained from an availability test), the results of the tank leaching test may be used to calculate an effective diffusion coefficient, D_e , for that component (often presented as pD_e , the negative logarithm of D_e). The ratio between the effective diffusion coefficient and the diffusion coefficient in water of a non-reactive component such as Na is called the tortuosity. The tortuosity is a measure of the physical retardation within the monolith (see e.g. de Groot and van der Sloot, 1992).

Most of the test methods used for this purpose, e.g. the Dutch standard tank leaching test NEN 7345 (NEN, 1994), are based on methods developed by the nuclear industry for the purpose of testing very hard and durable solidified (vitrified) low-level radioactive waste products, in parti-

cular ANSI/ANS-16.1 (ANS, 1986). As already mentioned, it appears that many solidified (“monolithic”) waste materials, which are landfilled, do not have such ideal properties.

The interpretation of a tank leaching test as described above is based on the assumption that the concentrations of contaminants in the water phase are kept very low throughout the duration of the test and, in particular, at no point in time approach equilibrium. The tests are designed to ensure that this requirement is fulfilled, although recent observations seem to indicate that this is not always achieved for all parameters (van der Sloot, 2004). In reference to a “real life” situation, these tank leaching tests simulate a scenario, which may be described as the leaching from a “pole in the middle of a river” (see Appendix 1). This is far from the situation in a landfill, and during the past few years it has become increasingly clear that the traditional tank leaching test may not be very useful in describing the leaching of contaminants from landfilled monolithic waste. The hydraulic situation in a landfill is further discussed in Chapter 7.

Under various assumptions, the results of (tank) leaching tests performed on monolithic materials, which usually take from a few days to 64 days, may be used to predict future leaching from the materials in various scenarios under conditions resembling those existing during the test. This can be done either by graphical extrapolation of the log-log curves (lines) of accumulated leached amounts vs. time combined with a mass balance or by combining effective diffusion coefficients with the results of availability tests (e.g. using Crank’s solution of Fick’s law of diffusion for a test body with semi-infinite dimensions, in which the initial concentration is uniformly distributed and the concentration on the surface between the test body and the eluate is constant with respect to time (Crank, 1975)). Even though tank leaching results cannot be evaluated properly without considering the results of the availability test, stabilised/monolithic materials are sometimes classified according to the pD_e values obtained (D_e in m^2/s). In the Dutch standard tank leaching test, a $pD_e > 12.5$ is interpreted as corresponding to a component with low mobility and a $pD_e < 11$ as corresponding to a component with high mobility. pD_e values between 11 and 12.5 are taken to correspond to components with average mobility (NEN, 1994).

The standardisation work carried out by CEN/TC 292 on leaching tests for monolithic waste is discussed in more detail in Appendix 1. As described in the Appendix, the realisation that the composition of the leachate from a “monolithic” landfill may be controlled by equilibrium rather than release rate (see also Chapters 2 and 3), has led to an attempt at developing a test procedure in which equilibrium between the (surface of the) monolithic material and the aqueous phase is attained in an initial step at low volume-to-surface-area ratio (L/A) and a long contact time

followed by a multi-step procedure under conditions similar to those in a “normal” tank leaching test (such as NEN 7345).

The results of the work, which was carried out by CEN/TC 292 WG 6: “Characterisation leaching tests” and in which DHI and VTT participated during this project, indicated that for most components, equilibrium conditions could not be attained as intended under conditions which were practical in terms of L/A and contact time. As a consequence, Working Group 2: “Compliance leaching tests” (WG 2) of CEN/TC 292 has made the following recommendation: “The compliance test for monolithic waste that is developed by WG2 should be split into two procedures consisting of a one step procedure (Part 1) and one that is derived (Part 2) from the start of the dynamic monolithic leaching test under development in WG 6. In addition, the standard should include a reference indicating that for assessment at equilibrium conditions there are existing tools such as the batch test for granular waste EN 12457 (Part 1 to 4) and the percolation test for granular material CEN TS 14405.” The first procedure (Part 1) was included primarily to meet a French regulatory requirement (see also section 9.3) and cannot be recommended, because it does not account for the leaching mechanism in its interpretation of the test results. The dynamic leaching test being developed by WG 6 is a characterisation test and has not yet been finalised. The test principle is still under discussion, but it is not unlikely that the test will resemble NEN 7343, when it is finalised. It is therefore also likely that this test will address situations in which the concentration of contaminants in the aqueous phase is controlled by the release rate rather than by equilibrium.

Some of the results produced by DHI and VTT as part of the CEN/TC 292 WG6 study are presented in Appendix 2.

4.2 Physical and chemical durability and integrity

The physical durability and integrity of a monolithic waste material is important, both in the short and long term. A tank leaching test can only be performed properly on a material which maintains its integrity and shape throughout the test period. And long term predictions about the leaching behaviour of the material after landfilling can only be based on the tank test results if the material maintains its integrity and basic properties for a sufficiently long period of time after landfilling. For a material which is hardened and/or compacted “in place”, the strength requirements may be moderate, whereas they should be more stringent for materials which are landfilled as individual blocks (still provided the leaching behaviour is predicted on the basis of a tank leaching test). The landfilled material must have a bearing capacity, which is sufficient to ensure the stability of the landfill during and after operation, and it must be able to withstand the effects of relevant freeze/thaw and wet/dry cycles. It may

be estimated that the maximum load on monolithic waste in a 20 m high/deep landfill is approximately 0.4 MPa. It is estimated that the typical strength of stabilised waste materials is 0.5 to 5 MPa, whereas normal concrete usually has strengths (far) exceeding 15 MPa (Laaksonen, 2004).

Chemical durability requirements would include resistance to any detrimental effects caused by reaction with e.g. water, carbon dioxide and oxygen. Excessive leaching of soluble materials (salts) may undermine the mechanical stability/physical integrity of a monolithic material, both for the purpose of testing and after landfilling.

4.3 Recommended test methods for monolithic waste

4.3.1 Testing hierarchy

As specified in the Annex II of the Landfill Directive (1999/31/EC), the general characterisation and testing of waste must be based on the following three-level hierarchy, which was developed by CEN/TC 292:

- Level 1: Basic characterisation. This constitutes a thorough determination, according to standardised analysis and behaviour-testing methods, the short and long-term behaviour and/or characteristic properties of the waste. A method to be used for subsequent compliance testing must be part of the basic characterisation.
- Level 2: Compliance testing. This constitutes periodical testing by simpler standardised analysis and behaviour-testing methods to determine whether a waste complies with permit conditions and/or specific reference criteria. The tests focus on key variables and behaviour identified by basic characterisation.
- Level 3: On-site verification. This constitutes rapid check methods to confirm that a waste is the same as that which has been subjected to compliance testing and which is described in the accompanying documents. It may merely consist of a visual inspection of a load of waste before and after unloading at the landfill site.

A particular type of waste must normally be characterised at level 1 and pass the appropriate criteria in order to be accepted on a reference list. In order to remain on a site-specific list, a particular type of waste must at regular intervals (e.g. annually) be tested at level 2 and pass the ap-

appropriate criteria. Each waste load arriving at the gate of a landfill must be subjected to level 3 verification.

4.3.2 Basic characterisation testing of monolithic waste (level 1 testing)

It is recommended that the waste to be tested is sampled representatively in a form which is representative of the form it is expected to have in the landfill. In many cases this may not be possible due to changes that will occur over a longer time period after the waste has been placed in the landfill. It may therefore be necessary to test the material in the form in which it appears at the time it is placed in the landfill, possibly after some specified stabilisation/maturation period under specified conditions. This is particularly important for monolithic waste which is placed in the landfill as a paste or a granular mixture which is subsequently compacted and/or left to harden.

The following properties should generally be tested as part of basic characterisation:

Total composition

The total composition of the monolithic waste material should be determined. The material should be crushed and finely ground (e.g. to < 0.125 mm) and subjected to chemical analysis for total content of major, minor and trace constituents. As a minimum, the total content of both the major constituents and the components for which leaching criteria have been set in Council Decision 2003/33/EC should be determined. If there is information that other inorganic or organic contaminants may be present at sufficiently high levels to pose a potential risk to the environment, they should be determined also.

Only methods providing true total contents should be applied. When using destructive analytical methods for inorganic constituents, appropriate total digestion methods such as EN 13565 (using HF, HNO₃ and HCl) should be used prior to ICP or AAS analysis rather than partial digestion methods such as EN 13657 (using aqua regia) or the Nordic method DS 259/SS 0228150/SFS 3044/NS 4770 (using nitric acid). For the determination of TOC, EN 13137 is recommended, and for other organics, appropriate methods must be applied.

Leaching as a function of L/S or time

If the reference situation is equilibrium controlled release (see Chapter 7), it is recommended that a representative sample of the monolithic waste material is crushed to a particle size < 4 mm and subjected to a column and a leaching batch test for inorganic components and DOC. The following tests should be used:

Column test: CEN/TS 14405: Characterisation of waste: Leaching behaviour tests – Up-flow percolation test (under specified conditions).

In this test, the waste is packed in a 30 cm long vertical column with a diameter of 5 cm (alternatively 10 cm), and demineralised water is passed through the material in up-flow. When the column is initially saturated, the flow is stopped for two days to allow for initial equilibrium, and then started again at a rate of 15 cm/24 hrs (slower, if needed to ensure local equilibrium in the column). 7 fractions of eluate corresponding to L/S = 0-0.1 l/kg, 0.1-0.2 l/kg, 0.2-0.5 l/kg, 0.5-1.0 l/kg, 1.0-2.0 l/kg, 2.0-5.0 l/kg and 5.0-10 l/kg are collected and analysed.

Batch test: EN 12457-1 to 3: Characterisation of waste – Compliance test for leaching of granular waste materials and sludges:

Part 1: One stage batch test carried out at a liquid to solid ratio (L/S) of 2 l/kg for materials with high solid content and with particle size below 4 mm (without or with particle size reduction). The waste is placed in a bottle with demineralised water and agitated for 24 hrs, then the eluate is filtered through a 0.45 µm filter and analysed chemically.

Part 2: One stage batch test carried out at a liquid to solid ratio (L/S) of 10 l/kg for materials with particle size below 4 mm (without or with particle size reduction). The waste is placed in a bottle with demineralised water and agitated for 24 hrs, then the eluate is filtered through a 0.45 µm filter and analysed chemically.

Part 3: Two stage batch test at liquid to solid ratios (L/S) of 2 l/kg and 8 l/kg for materials of with high solid content and with particle size below 4 mm (without or with particle size reduction). In each step, the waste is placed in a bottle with demineralised water. It is agitated for 6 hrs in the first stage and 18 hrs in the second stage before the eluates are filtered through 0.45 µm filters and analysed chemically.

The choice between Part 1, Part 2 and Part 3 generally depends on national preferences.

The eluates from both the column test and the batch test should be analysed for major inorganic components as well as minor and trace inorganic components (as a minimum those for which leaching criteria have been set) and DOC. pH, conductivity and redox potential (optional) are determined as part of the leaching procedures.

If the reference situation for a given monolithic waste material in a landfill can be shown to be release rate control (see Chapter 7), and a satisfactory relationship can be established between the result of a tank leaching test and the risk of groundwater or surface water pollution allowing the development of associated acceptance criteria, a tank leaching

test might be applicable. As described above and in Appendix 1, no European standard tank leaching test is yet available. Until a European standard tank leaching test becomes available, it is recommended to use the Dutch standard tank leaching test NEN 7345: "Leaching characteristics of soil and stony building and waste materials. Leaching tests. Determination of leaching of inorganic components from building and monolithic waste materials with the diffusion test." In this test, a monolithic specimen is immersed in demineralised water at a volume to surface area (L/A) ratio of 8 m³/m². The water is exchanged with clean demineralised water at increasing time intervals to obtain a total of 8 fractions of eluate over a period of 64 days. The eluates are filtered through 0.45 µm filters and analysed chemically. If it is desired to determine an effective diffusion coefficient, the procedure includes an availability leaching test.

Leaching as a function of pH

The sensitivity of the leachability of monolithic waste to changes in pH should be determined on representative samples which are crushed and finely ground (< 1 mm is required, but < 0.125 mm is recommended). Either of two CEN methods may be recommended:

prEN 14997: Characterisation of waste: Leaching behaviour tests – Influence of pH on leaching with continuous acid/base addition.

and

prCEN/TS 14429: Characterisation of waste: Leaching behaviour tests – Influence of pH on leaching with initial acid/base addition.

Both tests are expected to be ready as CEN technical specifications (rather than standards, ENs, since they have not been validated) around the end of 2005. In both tests, a series of batch equilibrium tests are carried out at L/S = 10 l/kg with a contact time of 48 hrs at various pH values ranging from 4 to 12 (usually 8 different pH values). In prEN 14997, pH is controlled continuously using automated feed-back addition of acid or base to maintain the desired pH. In prCEN/TS 14429, the acid or base is added during the first part of the test based on a pre-titration of the material. The eluates are filtered through 0.45 µm filters and analysed chemically. Both tests also provide an alkalinity/acidity curve for the material.

The eluates should be analysed for major inorganic components as well as minor and trace inorganic components (as a minimum those for which leaching criteria have been set) and DOC.

4.3.3 Compliance testing of monolithic waste (level 2 testing)

For compliance testing of monolithic in the equilibrium controlled landfilling situation, one of the batch leaching procedures EN 12457-1 to 3 may be recommended.

If deemed necessary, the column leaching procedure EN 14405 may also be used as a compliance test, possibly in a simpler version, where some or all eluate fractions are combined prior to analysis.

The eluates should be analysed for those components for which criteria have been set.

No recommendations are given for compliance testing in the release rate controlled situation.

4.3.4 On-site verification (level 3 testing)

No testing is recommended at this level.

5. The TAC methodology for development of acceptance criteria

The criteria and limit values for acceptance of “monolithic” waste at landfills for non-hazardous and hazardous waste landfills must provide (at least) the same level of environmental protection as given by the corresponding criteria for granular waste. It will therefore be reasonable to follow the same basic procedure as the one used by the TAC Subcommittee on the Landfill Directive to set criteria for granular waste in relation to groundwater protection with appropriate adjustments to account for the “monolithic” nature of the waste. This chapter describes the stepwise procedure used by the TAC Subcommittee. (Hjelmar et al. 2001, Hjelmar 2003, DHI & ECN 2004).

5.1 Outline of the procedure

The procedure used to set acceptance criteria for landfilling of granular waste consisted of a series of consecutive steps. 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 concentra-

tion 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:

- a) Choice of primary target(s) and principles
- b) Choice of critical parameters and primary criteria values
- c) Description of the landfill scenario
- d) Description of the environment scenario
- e) Description of the source of potential contamination
- f) Description and modelling of the migration of the contaminants from the landfill to the POC(s)
- g) Determination of attenuation factors
- h) Application of the results to criteria setting (“backwards” calculation)
- i) Transformation of the source term criteria to limit values at different L/S values

The impact calculation is carried out using three coupled source and transport models in series: one (model 1) describing the source of contamination (the production of leachate from the landfill), one (model 2) describing the transport of contaminants in the unsaturated zone and one (model 3) describing the transport of contaminants through the saturated zone to the POC. Model 2 and model 3 would be the same for granular and monolithic waste irrespective of the source model used, whereas model 1 might have to be adjusted, if the source description for the monolithic waste is different from that for the granular waste. The principle of the three connected models is illustrated in figure 5.1.

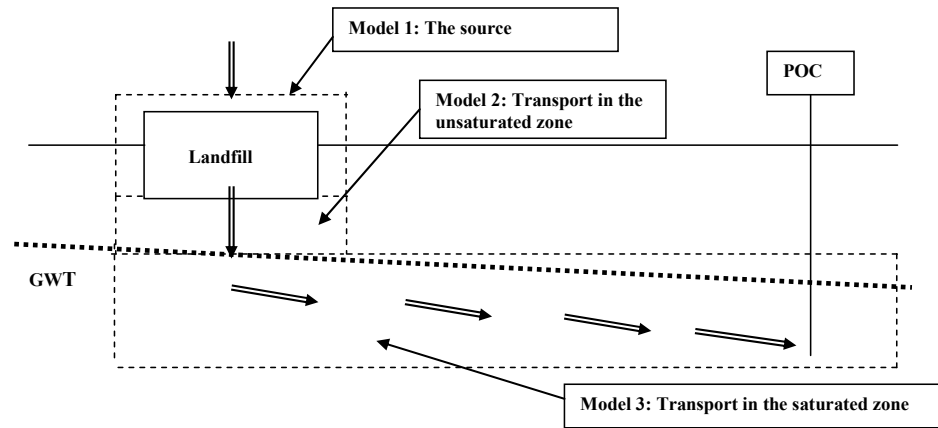


Figure 5.1 Cross-section showing the principle of three coupled source and transport models used for the forward calculation at a landfill scenario.

It should be kept in mind that this procedure is aiming at the development of general (European) acceptance criteria and that it therefore necessarily includes generalisations that, from a local point of view, often may seem ill-fitting and unrealistic. The same methodology may, however, be used for site-specific impact assessments if a number of the generalised parameters are substituted by parameters that better fit the local situation.

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

The major potential impact on the environment from a landfill, both in the short and longer term, 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 in most cases 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. In the TAC calculations, the POC chosen was located 20 m downstream of the edge of the landfill, except for the very mobile components chloride and sulphate, for which a POC located 200 m downstream of the edge of the landfill was chosen. At this point, the condition to be considered was the highest average annual concentration of a given component resulting at any time from the leachate discharged from the landfill. The concentration level was averaged over the entire depth of the aquifer.

5.3 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 is likely to provide regional groundwater quality criteria within the EU.

In the TAC calculations, the existing drinking water criteria, with modifications of some of those parameters, which are very high compared to normal groundwater values, was used. The EU Drinking Water Directive (EU, 1998) and WHO drinking water criteria (WHO, 1996) 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⁻, NH₄⁺, NO₃⁻, NO₂⁻ and SO₄²⁻. The final selection of parameters to be regulated was a compromise between the availability of leaching results, the existence of water quality criteria and the need for regulation as seen by the EU member states participating in the work of the TAC. The components for which leaching criteria were developed are listed in table 5.1.

Table 5.1 Groundwater quality criteria used at the POC in the TAC calculations (Hjelmar & Holm, 2004).

Parameter	Groundwater quality criteria set at POC (mg/l)	Comments
As	0.010	WHO (1996)
Ba	0.700	WHO (1996)
Cd	0.004	Set by the TAC (WHO (1996) = 0.003)
Cr	0.050	WHO (1996)
Cu	0.050	Lowered by the TAC (WHO (1996) = 2)
Hg	0.001	WHO (1996)
Mo	0.070	WHO (1996)
Ni	0.020	WHO (1996)
Pb	0.010	WHO (1996)
Sb	0.005	WHO (1996)
Se	0.010	WHO (1996)
Zn	0.100	Lowered by the TAC (WHO (1996) = 3)
Chloride	250	WHO (1996)
Fluoride	1.5	WHO (1996)
Sulphate	250	WHO (1996)
DOC	10	Set by the TAC
Phenol index	0.100	Set by the TAC (only for landfills for inert waste)

5.4 Step 3: Description of the landfill scenario

This step involves a detailed description of the climatic physical appearance and properties of the landfill and the waste. This is part of the source term calculations, which are described in detail in section 7.2. Together with climatic and hydraulic data this 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 release rate) controlling the release of contaminants. This forms the basis for the selection of appropriate leaching tests in step 5. When the methodology is used for the setting of general limit values, the scenario should be fairly “typical” and relatively simple. A simplified plug flow description is applied and conditions such as edge effects and preferential flow are generally not accounted for.

5.5 Step 4: Description of the environment scenario

The environmental scenario is intended to provide a simplified description of the characteristics of a “typical” landscape in which the landfill 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). It should also include estimates of the retention properties of the saturated and unsaturated zones (for the purpose of modelling expressed in terms of K_d values).

The description of the environment scenario, mainly in terms of the climatic conditions and the size and properties of the unsaturated and saturated zones as used in the TAC calculations are presented in table 5.2. The dispersivities in the saturated zone were set relatively high in order to force the full vertical mixing, which had been assumed by the TAC.

In the modelling, retardation in the unsaturated and saturated zones was incorporated assuming a linear equilibrium-sorption relationship, K_d , between the concentration C_s of the component in the soil (mg/kg) and the concentration C_w of the component in the water phase (mg/l). Table 5.3 presents the values of the distribution coefficient $K_d = C_s/C_w$ used in the TAC calculations.

Table 5.2 Parameter values used in the model calculations of transport in the unsaturated and saturated zones in the TAC calculations (DHI & ECN, 2003, DHI & MST, 2005).

Parameter	Unit	Values used by the 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
General net rate of infiltration	mm/year	300
Thickness of unsaturated zone (clay bottom liner)	m	1 m (non-haz) 5 m (haz)
Dispersivity of the unsaturated zone	m	0
Aquifer/saturated zone properties:		
Thickness of aquifer	m	Approx. 6
Upper boundary	-	Closed
Fixed hydraulic head at downstream boundary	m	Approx. 4.1
Horizontal hydraulic conductivity $K_x = K_y$	m/s	1.4×10^{-4}
Vertical hydraulic conductivity K_z	m/s	1.4×10^{-4}
Effective porosity	-	0.3
Longitudinal dispersivity	m	20
Transversal dispersivity	m	4
Vertical dispersivity	m	2

Table 5.3 Values of K_d used in the TAC calculations (DHI & ECN, 2003, Hjelmar et al., 2005).

Parameter	Values of K _d used in the TAC calculations
	l/kg
As	50
Ba	2
Cd	20
Cr	100
Cu	14
Hg	1
Mo	10
Ni	50
Pb	50
Sb	5
Se	5
Zn	30
Chloride	0
Fluoride	2
Sulphate	0
DOC	0
Phenols	40*

* The K_d value for phenols was assumed to incorporate some biological degradation, although degradation in general was not included in the modelling.

5.6 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 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/non-equilibrium situation.

The source term under solubility/equilibrium-like conditions is described in detail in section 7.4. Under these conditions, the description of the source term and the subsequent checking of the compliance of a given type of waste with the acceptance criteria are based on the results of percolation related leaching tests in which equilibrium (or local equilibrium) is assumed, i.e. column or batch leaching tests, where the results may be described as a function of L/S. Another approach, in which the modelling and compliance are based on the results of a dynamic non-equilibrium test (a tank leaching test) is discussed in section 9.1 (the approach proposed by the UK). However, since this approach does not account for the build-up of contaminant concentrations in the bulk water phase and the resulting solubility control of the release, nor for the possible effect of

availability limitations, it seems to lead to unrealistically stringent acceptance criteria.

5.7 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 landfill to the groundwater and the POC(s), the contaminants leached from the inert waste are first transported vertically through the unsaturated zone 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 (low K_d values) and only affected by dilution/dispersion, whereas others (e.g. lead) are almost immobile, even over longer periods of time (high K_d values). 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 the peak initial 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 landfill are relevant, and the peak concentration in the groundwater near the landfill generally shows a less straightforward relationship to the peak concentration in the leaching test than it does 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.

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, see table 5.4 (Hjelmar et al. 2001). 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.

In the TAC, forward release and transport modelling for granular waste was performed in parallel both by the UK, France and the Netherlands and Denmark (Hjelmar et al., 2001). The input to the transport model for the unsaturated zone in terms the flux of each contaminant as a function of time was calculated for each type of landfill by combining the information on the flow of leachate out of the landfill (section 7.4.3) with the information on the composition of the leachate as a function of L/S (section 7.4.4). For a given scenario, the relationship between L/S and time may be calculated as shown in section 7.4.5. In the TAC calculations DHI and ECN used CXTFIT/ECOSAT and HYDRUS 2D for the unsaturated zone and MODFLOW and MT3D for the saturated zone (DHI and ECN, 2003). More recently, in connection with the Danish implementation of Council Directive 2003/33/EC, a numerical 3 D flow and transport code, MIKE-SHE, in which the modelling of the transport through the unsaturated and the saturated zones is integrated, has been applied by DHI for similar contaminant transport calculations (Hjelmar et al., 2005).

Table 5.4 Overview of transport models applied to the scenarios (Hjelmar et al. 2001).

Model	Main features	Waste release	Unsaturated zone	Soil interaction	Ground-water hydro-logy	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

Both in the TAC calculations and in the Danish implementation calculations contaminant/subsoil interactions were taken into account by inclusion of simple reversible sorption processes, assuming they could be described by linear sorption isotherms (expressed in terms of K_D values for each contaminant, see table 5.3), both in the unsaturated and the saturated zones. Linear adsorption is included in most up-to-date groundwater transport models.

5.8 Step 7: Determination of attenuation factors

Based on “forward” modelling of the release and subsequent transport through the unsaturated zone and the aquifer of each component of interest, the peak concentration, C_P , resulting at any time in the groundwater at the POC is determined for each component. The attenuation factor f_a , which is an expression of the dilution resulting from the transport through the unsaturated and saturated zones, is calculated as the ratio between the maximum concentration of the component in leachate at the base of the landfill and C_P :

$$f_a = C_0/C_P \quad (5.1)$$

In the TAC calculations for all three types of landfills, the peak concentrations C_P in the groundwater at the POC (20 m downstream of the landfill) typically occurred more than 2000 years after the start (and the closure) of the landfill. One attenuation factor is determined for each component in each scenario (e.g. inert waste landfill, non-hazardous waste landfill and hazardous waste landfill).

5.9 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 for mineral-like granular waste under equilibrium-like conditions. Groundwater quality criteria (C_{GQC}) are set at the POC as described in section 5.3 and table 5.1. The maximum allowable concentration, C_0 , in the leachate at the base of the landfill may then be calculated from the attenuation factor and C_{GQC} :

$$C_0 = f_a (C_{GQC} - C_{BG}) \quad (5.2)$$

where C_{BG} is the background concentration of the component in question in the groundwater (the model calculations were performed using $C_{BG} = 0$ in the saturated zone).

In the TAC calculations, a value of $C_{BG} = 0$ was used for all components in equation (5.2) which then reduces to

$$C_0 = f_a C_{GQC} \quad (5.3)$$

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

When C_0 corresponding to the groundwater quality criteria, C_{GQC} , has been determined for a given contaminant, equation (7.3) in section 7.4.4 can be used to calculate the cumulative amount, E , of that component that has been leached per kg of granular waste under (local) equilibrium-like conditions during the period it takes for L/S to increase from 0 to a given value:

$$E = (C_0/\kappa) (1 - e^{-(L/S)\kappa}) \quad (7.3)$$

The leaching limit values corresponding to results of column or batch leaching tests expressed in terms of leached amounts at $L/S = 2$ l/kg and $L/S = 10$ l/kg, respectively, may then be calculated as follows:

$$E_{L/S=2} = (C_0/\kappa) (1 - e^{-(2 \text{ l/kg})\kappa}) \quad (5.4)$$

$$E_{L/S=10} = (C_0/\kappa) (1 - e^{-(10 \text{ l/kg})\kappa}) \quad (5.5)$$

where C_0 is calculated from equation (5.2) or (5.3) and κ is the component-specific kinetic constant obtained from table 7.9. κ is actually also

waste-specific, but was, due to limited availability of data (and for the sake of simplicity), considered waste-independent in the TAC calculations.

The limit values calculated above may e.g. refer to the accumulated leached amounts at $L/S = 2$ l/kg and $L/S = 10$ l/kg in the percolation (column) leaching test CEN/TS 14405. Limit values could be calculated for any other L/S value for which leaching data would be available. The values also refer to the results of the compliance (batch) leaching test EN 12457-1 ($L/S = 2$ l/kg), EN 12457-2 ($L/S = 10$ l/kg) and EN 12457-3 ($L/S = 2$ and 10 l/kg).

The maximum allowable concentration, C_0 , may also refer to the highest concentration occurring in column leaching test. For most (but not necessarily all) components the highest concentrations will appear in the initial eluate fraction. In the Council Decision 2003/33/EC, C_0 has been set as a limit value referring to the concentration of components in the first eluate fraction ($L/S = 0.0 - 0.1$ l/kg) in the column leaching test CEN/TS 14405.

6. Monolithic landfills in practice

A survey has shown that no monolithic landfills currently are in use or planned in Iceland. In Denmark, Kommunekemi A/S annually landfills approximately 300 tonnes of monolithic waste in the form of used batteries (127 tonnes) that are mixed with cement (173 tonnes) and placed in 200 kg steel drums in a landfill with artificial top and bottom liners to prevent contact with water. A few tonnes per year of Be- and Hg-containing waste are landfilled in a similar manner (Kjærsgaard, 2004). Norway may have two landfills for monolithic sites (one for stabilised hazardous waste at Langøya and one for industrial waste from the titanium industry). Finland has one actual landfill for monolithic waste (cement-stabilised contaminated soil) and several storage fields for the same type of material (Appendix 3).

Sweden, on the other hand, has several landfills for monolithic waste (Appendix 4). Most of these landfills are dedicated to various types of stabilised municipal solid waste incinerator (MSWI) fly ash or acid gas cleaning residues. Landfilling of stabilised fly ash as monolithic waste is generally practised in Sweden. The fly ash is mixed with sludge-water or other water and the slurry is landfilled (according to the so-called Bamberg stabilisation model or after cement stabilisation, (Wänstedt, 2001)). The slurry is either driven to the landfill by truck (or deposited directly in a slope, e.g. Boden Energi AB), or pumped directly to the landfill. The slurry is left to harden in the landfill. At some landfills the slurry is compacted after being deposited (example Högbytorp), while the only compaction occurring at others is that resulting from the necessary driving on the surface (example Boden). Landfilling of stabilised MSWI APC residues has been confirmed for Sofielund, Högbytorp, Boden Energi and Renova, but the survey has not been complete and more landfills with stabilised MSWI APC residues are likely to exist in Sweden. Other types of monolithic landfills in Sweden includes two examples regarding soil treatment, one example where metal contaminated soil was mixed with lime, and one example where residue from soil washing was mixed with cement.

With the exception of the limited amounts of heterogeneous mixes of batteries and Be- and Hg-containing waste and cement in Denmark, there are no reports on landfilling of pre-fabricated monolithic blocks of stabilised waste in the Nordic countries. Information obtained at a project workshop with participants from the Nordic countries, the Netherlands, UK and Spain seems to indicate that most “monolithic” landfills in Europe receive waste, which is compacted or hardened after being placed in the landfill, rather than pre-fabricated blocks of waste (Baun, 2003). This

is also the case at the large monolithic waste landfill in Rotterdam in the Netherlands, where a reduction in leachate production to 4 – 15 % of that produced at an adjacent empty landfill compartment was observed (van Zomeren et al., 2003). This is ascribed to increased evapotranspiration and surface run-off, but initial uptake of water in the waste could also be a contributing factor. Field observations have indicated that leachate produced at the site was in equilibrium with the waste in the landfill (van der Sloot, 2004).

A landfill of 1 m³ blocks of cement stabilised MSWI APC residues was established in Switzerland, but some years later it was observed that the blocks were crumbling (Johnson, 2000). In a study in the UK, metal plating waste was stabilised with Portland cement and coal fly ash in the form of 2.5 m³ cylindrical blocks and exposed to the environment for 10 years (Fitch & Cheeseman, 2003). A substantial disintegration of the stabilised material was observed, particularly near the surface.

Most of the above information supports the notion that very few landfills contain truly “monolithic” (hard, concrete-like, coherent) waste in an environment where the composition of the leachate is controlled by the rate of release of contaminants from the solid phase.

7. Description of the source term when assuming equilibrium

7.1 Scenario description and equilibrium criterion

In Appendix 5 a simplified release model has been developed for a monolithic landfill scenario, and a relatively simple criterion has been set up to determine whether or not the concentration of the various contaminants in the leachate at the bottom of the landfill is controlled by equilibrium/solubility.

Two situations are considered to determine the relative influence on the leaching process of availability, solubility and kinetics of chemical and physical processes:

In the first situation, the velocity of the infiltrating water in the fractures of the landfilled material is assumed to be slow compared to the kinetics of the chemical reactions and the diffusion process for a given component. This makes the local equilibrium assumption valid for that component. The aqueous phase and the solid phase are in equilibrium and the concentration in the matrix pore water is equal to the concentration of the component in the fracture domain. The equilibrium concentration may be controlled either by availability or solubility. As the controlling minerals are exhausted, a reaction front will develop and travel downwards.

In the second situation, the velocity of the infiltrating water in the fractures is fast compared to the kinetics of the chemical reactions and the diffusion processes. This will create a non-equilibrium situation, and diffusion and chemical reaction rates become significant controlling mechanisms. In this case the concentration in the matrix pore water is a function of the distance from the surface of the fracture, the depth and time. In contrast to the equilibrium situation, reaction fronts will develop from the fracture and travel horizontally into the matrix.

The simplified model may be applied to a monolithic landfill scenario to determine which of the two situations is likely to prevail under specified conditions.

The scenario used for this purpose is a landfill filled to a height H (m) with monolithic waste of a geometric surface area per unit volume A (m^2/m^3). The rate of infiltration of precipitation through the top cover of the landfill (= the rate of production of leachate at the bottom of the landfill) is INF (mm/year). C (mg/l) is the concentration of a given component in the leachate in the fracture domain at the bottom of the landfill at a given time after the infiltrating water has reached this point. C_{eq} (mg/l)

is the solubility limit of the component in question. k ($\text{mg}/\text{m}^2/\text{day}$) is the rate coefficient for the assumed zero order release (i.e. constant rate of release) of the component from the surface of the fracture into the water present in the fracture. As derived in Appendix X1, the concentration of the component in the leachate at the bottom of the landfill can be expressed in terms of C_{eq} , A , H , k and INF :

$$C = C_{\text{eq}} (1 - e^{-365 k A H / (\text{INF} C_{\text{eq}})}) \quad (7.1)$$

From the above and Appendix X1 it follows that the conditions under which equilibrium/solubility controlled release can be assumed are fulfilled when the value of C approaches C_{eq} , which corresponds to C/C_{eq} approaching 1.

7.2 Factors influencing the release controlling mechanism

The values of C_{eq} , k , A , H and INF determine whether or not C approaches C_{eq} for a particular component and a particular waste in a specified scenario. These parameters are discussed in more detail below.

7.2.1 Solubility, C_{eq}

The solubility C_{eq} of a specific component depends on the component in question and the nature of the solid phase as well as on several other factors, including temperature, pH, ionic strength and DOC. As C approaches C_{eq} , the driving force for the release will decrease and disappear totally for $C = C_{\text{eq}}$, i.e. chemical equilibrium will exist between the component present in dissolved form in the leachate and in the solid phase. Estimates of C_{eq} for specific materials and specific components may e.g. be obtained from pH-static leaching tests. According to the equation above, a lower value of C_{eq} favours equilibrium conditions.

7.2.2 Release rate, k

The constant release rate, k , represents a crude approximation to the actual situation and it is therefore important both to investigate the sensitivity of the equilibrium conditions to variations in k and to try to find and use realistic estimates of k values. Estimates of k values or the order of magnitude of k for specific materials and specific components may possibly be obtained from the results of tank leaching tests. In the case of diffusion control, the release kinetics are half order rather than zero order. A higher value of k favours equilibrium conditions and low values of k

may therefore be critical with respect to fulfilling the requirements of equilibrium control.

7.2.3 Geometric surface area per unit volume, A

The geometric surface area per unit volume, A , describes the surface-to-volume ratio, i.e. the surface area (in contact with water/leachate) per volume of the landfilled material. A depends on geometry and scale, and is also a measure of the relative amount of crack surfaces in the landfilled material. For a cube with side length = a and n sides exposed to water (and no cracks), $A = n/a \text{ m}^2/\text{m}^3$ (i.e., $A = 12 \text{ m}^2/\text{m}^3$ for a cube with 0.5 m sides that are all wet). For a monolithic landfill with 2 vertical cracks from top to bottom per running meter in one direction only, A will be approximately $4 \text{ m}^2/\text{m}^3$. It has not been possible to find any actual estimates of A for monolithic landfills. Larger values of A will favour equilibrium conditions at the bottom of the landfill.

7.2.4 Height, H

The height, H , describes the average height of the landfill. In the TAC calculations $H = 20 \text{ m}$ was used for all three categories of landfills. The likelihood that equilibrium conditions will control the composition of the leachate at the bottom of the landfill increases with increasing height, since this increases the retention time for the water in the fractures.

7.2.5 Rate of infiltration, INF

The net rate of infiltration, INF , through the top cover into and through the landfill plays an important role in determining whether or not equilibrium conditions will prevail. In the TAC calculations, the infiltration through the top cover was assumed to vary between 0 and 200 mm/year for non-hazardous and hazardous waste landfills (see table 7.8 in section 7.4.3). The Landfill Directive does not specify any direct requirements on the rate of infiltration. It is most likely that the relatively low permeability that can be expected for monolithic waste will create a larger surface or subsurface run-off than is normally seen for granular waste. Apart from that, the rate of infiltration will depend on climate and on national or local requirements on the performance of the top cover. A lower rate of infiltration favours the establishment of equilibrium conditions.

7.2.6 Relative influence of parameters on the equilibrium conditions

Figures 7.1, 7.2 and 7.3 illustrate the relative influence of the various parameters on the contaminant release conditions. The parameter values chosen are not necessarily realistic for real waste and landfills, and the

figures are only meant to demonstrate the interrelationship between the parameters

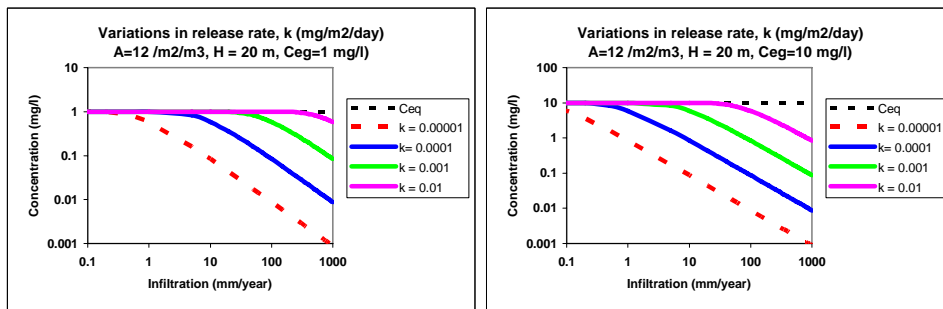
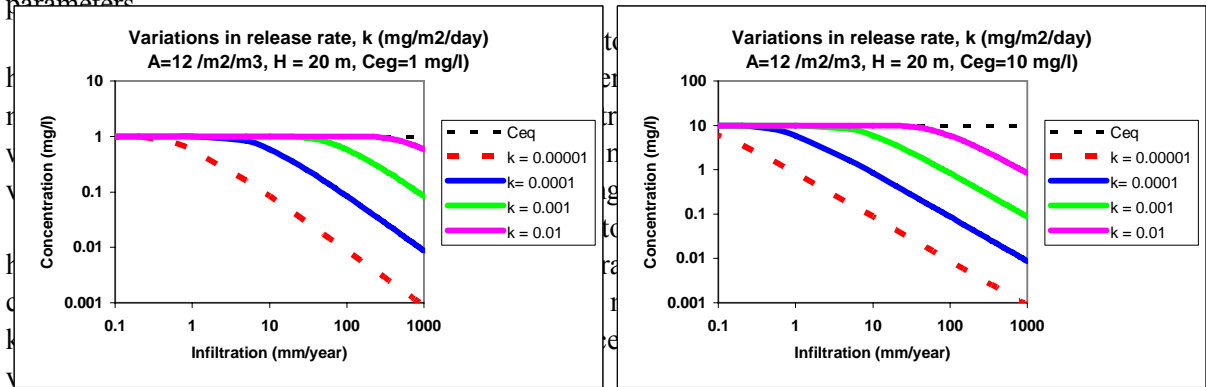


Figure 7.1 The concentration C at the bottom of a monolithic landfill as a function of the rate of infiltration for $C_{eq} = 1$ and 10 mg/l for various values of the release rate k .

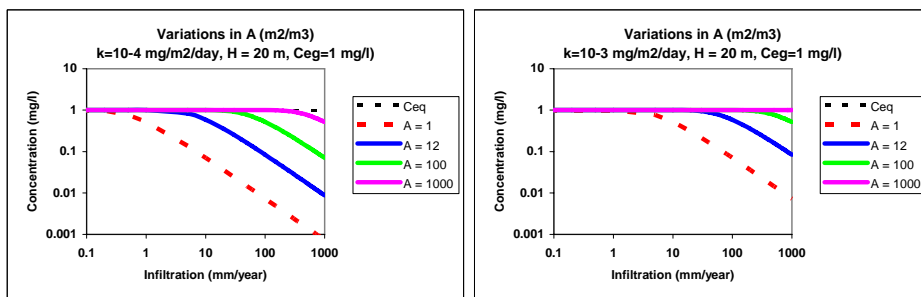


Figure 7.2 The concentration C at the bottom of a monolithic landfill as a function of the rate of infiltration for $C_{eq} = 1$ mg/l and $k = 10^{-4}$ and 10^{-3} mg/m²/day for various values of the geometric surface per unit volume (A).

In figure 7.3 the concentration C , at the bottom of a landfill with a geometric surface area per unit volume of $12 \text{ m}^2/\text{m}^3$ is presented as a function of the rate of infiltration for a component with an equilibrium concentration of 1 mg/l and release rates, k , of 10^{-4} and $10^{-3} \text{ mg/m}^2/\text{day}$ for landfill heights ranging from 5 to 30 m .

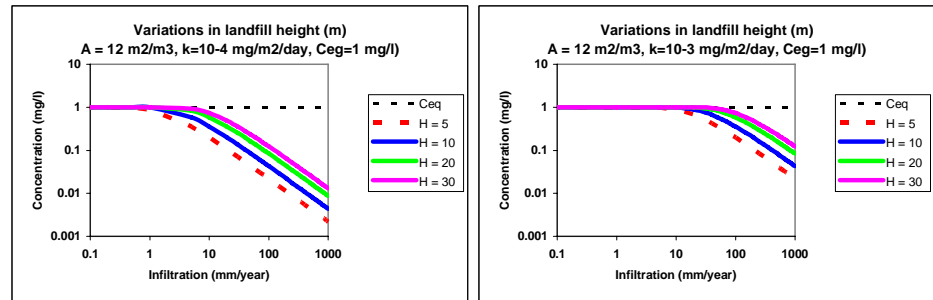


Figure 7.3 The concentration C at the bottom of a monolithic landfill as a function of the rate of infiltration for $C_{eq} = 1$ mg/l and $k = 10^{-4}$ and 10^{-3} mg/m²/day for various landfill heights (H).

The figures show how equilibrium conditions (where C is equal or close to C_{eq}) are favoured by increasing values of k , A , and H and by decreasing values of C_{eq} and Inf .

7.3 Testing the equilibrium condition with actual waste data

7.3.1 Materials and data sources

In order to validate (or reject) the equilibrium assumption, the model should be tested using values of C_{eq} , k , A , H and INF , which are representative of actual monolithic waste and actual landfill conditions. As already indicated in section 7.2, such data are not easily available. However, during this project both DHI and VTT have participated in a study of a tank leaching test (see also section 4.1 and Appendix 2) organised by CEN/TC 292/WG6. In this study, tank leaching results were produced on three materials. Two of these materials, StabW and BRC-1, were stabilised wastes, and therefore of potential interest in this context. DHI also performed a pH-static leaching test on one of the materials (StabW). For the other material (BRC-1), pH-dependent leaching test results and column leaching test results (on crushed material) have been provided by ECN in the Netherlands (van der Sloot, 2004). In addition, ECN has provided results of the Dutch leaching test (NEN 7345) and a pH-static leaching test performed on a third material, a cement stabilised waste (BRC-2). In the tank leaching tests, all three materials were tested as monolithic test bodies. In the pH-dependence leaching tests, the test bodies were finely ground, and in the column test, the monolithic material was crushed to a granular material (< 4 mm).

The three monolithic materials may be characterised as follows:

StabW: Air pollution control (APC) residue from fluidised bed combustion (FPC) of refuse derived fuel (RDF) stabilised/solidified with pozzolanic fly ash and a little ballast (bottom sand from a PFBC coal fired power plant). The reported content of chloride in the APC residue (60 to 70 g/kg) is lower than that normally found in acid gas cleaning residues and fly ash from MSW incinerators (Nyholm, 2005). The material was produced by Ragn-Sells AB in Sweden and is an example of stabilised waste for landfilling.

BCR-1: Stabilised mixture of MSWI fly ash (9.7 % by weight), Portland cement (24.9 % by weight), natural sand (25.8 % by weight), silver sand (25.8 % by weight) and water (13.8 % by weight). Density: 1968 kg/m³, permeability: 4.9×10^{-12} m/s. This material was produced for testing purposes by ECN. It contains a relatively small proportion of actual waste material (MSWI fly ash) and probably more resembles a product for utilisation than a waste for landfilling.

BCR-2: MSWI FA residues stabilised with cement (5 – 10 %) and a proprietary stabilisation agent. This material was produced for testing purposes by ECN and is probably a realistic representative of stabilised waste for landfilling.

The various tests carried out on the three monolithic waste materials and used in this report are summarised in table 7.1. Some of the results of the tank leaching tests and pH dependent leaching tests (the pH-static test and the ANC test) are presented in Appendix 2.

Table 7.1 Tests from which the results are used in this report. The eluates from the tests performed by DHI were analysed by AGH in Krakow, Poland.

Material	Form in test	Test methods	Test lab
StabW	Monolithic	Experimental tank leaching procedure	DHI
	Finely ground	pH-static leaching test (prEN 14997)	DHI
BCR-1	Monolithic	Experimental tank leaching procedure	DHI
	Finely ground	ANC leaching test (prCEN/TS 14429)	ECN
	Crushed to < 4 mm	Column leaching test (CEN/TS 14405)	ECN
BCR-2	Monolithic	Dutch standard tank leaching test (NEN 7345)	ECN
	Finely ground	pH-static leaching test (prEN 14997)	ECN

If it is assumed that equilibrium conditions are achieved in a pH dependent leaching test, the equilibrium concentration, C_{eq} , may be estimated from results of this test. In the pH-static test (prEN 14997) acid or base is

added continuously by feed-back control to a stirred vessel containing finely ground material and demineralised water at $L/S = 10$ l/kg to maintain a constant, pre-set pH. After a contact period of 48 hours the eluate is filtered through a $0.45 \mu\text{m}$ filter and analysed chemically. This is done for a range of pH values between 4 and 12. In the ANC test (CEN/TS 14405), the amount of acid or base needed to achieve different pH values in the range of 4 to 12 is determined in a preliminary titration step. This amount of acid/base is then added to the material with the leachant (demineralised water) in three steps during the first two hours to achieve $L/S = 10$ l/kg. After 48 hours of agitation, the eluate is filtered through a $0.45 \mu\text{m}$ filter and analysed chemically.

The equilibrium concentrations determined from the test vary widely as a function of pH for many components, and it is therefore chosen to use the highest concentration found in the pH interval between approximately 8 and the material's own pH (which is between 10 and 13 for the materials in question). This is a conservative choice in relation to establishing the fulfilment of the equilibrium assumption, since a higher C_{eq} makes it more difficult to fulfil the assumption. The lower end of the interval, $\text{pH} = 8$, seems safe since all three materials have high acid neutralisation capacities and are very unlikely to be acidified beyond $\text{pH} = 8$ for a long period of time when exposed to external influences under landfill conditions. C_{eq} is likely to decrease with time for most components as dissolved matter is gradually removed from the system with the leachate.

A tank leaching test such as NEN 7345 and the last several stages of the procedure tested by CEN/TC 292/WG6 provides a measure of the flux (the rate of release e.g. in terms of $\text{mg}/\text{m}^2/\text{day}$) as a function of time for a given material and a given component under optimum release conditions. If the release is diffusion controlled, it will decrease as a function of time (as a linear function with a slope of -0.5 in a log flux-log time graph). A build-up in concentration of a component in the water phase will tend to reduce the release rate for that component. Using tank leaching data to estimate the release rate k in determining if the equilibrium assumption is fulfilled is thus not conservative with respect to k , since a lower k will favour non-equilibrium. To be as conservative as possible when using the tank leaching data, the (lowest) flux, usually from the last step of the tank leaching procedure (in most cases determined from day 36 to day 64 or day 37.25 to day 65.25) is used as the estimate of k .

The values of C_{eg} chosen for the scenario calculations are presented in table 7.2, which also shows at which pH they were determined. The values of k chosen for the scenario calculations are shown in table 7.3 together with the period during which they were determined. Only results for components for which the leaching is regulated in Council Directive 2003/33/EC are shown. Results were not available for all components.

Table 7.2 The values of C_{eq} chosen for the scenario calculations from the results of the pH dependence leaching tests shown in Appendix 2. The table also shows the pH values at which the equilibrium concentrations have been determined (the highest value between approximately 8 and the materials own pH was chosen).

Component	StabW		BCR-1		BCR-2	
	C_{eq} (mg/l)	pH	C_{eq} (mg/l)	pH	C_{eq} (mg/l)	pH
As	0.041	7.96				
Ba	0.59	7.96	1.1	11.00	1.9	8
Cd	0.0025	7.96	0.29	7.50	1.8	8
Cr	0.099	7.96	0.92	9.96	0.41	10
Cu	0.0041	7.96	0.048	7.50	0.016	12
Hg	0.0019	7.96				
Mo	0.10	7.96	0.10	9.96	0.77	12
Ni	0.013	7.96	1.5	7.5	0.16	8
Pb	0.0015	7.96	0.26	12.55	1.9	12
Sb	0.26	7.96	0.08	7.5	0.49	10
Se	0.059	7.96				
Zn	0.2	8.93	2.3	7.5	5.7	8
Chloride	2000	7.96			5400	8
Fluoride						
Sulphate	2000	10.00	1200	10.00	3200	12
DOC						

Table 7.3 The values of k chosen for the scenario calculations from the results of the tank leaching tests shown in Appendix 2. The table also shows during which period of the tank test the release rates have been determined (the lowest value, which usually occur during the final period, was chosen).

Component	StabW		BCR-1		BCR-2	
	k	Period	k	Period	k	Period
	mg/m ² /d	days	mg/m ² /d	days	mg/m ² /d	days
As	0.012	37 – 65	0.0040	37 – 65		
Ba	0.60	37 – 65	0.12	37 – 65	0.56	36 – 64
Cd	0.00054	18 – 37	0.00043	18 – 37	0.0018	36 – 64
Cr	0.043	37 – 65	0.042	37 – 65	0.10	36 – 64
Cu	0.0018	37 – 65	0.0029	37 – 65	0.0026	36 – 64
Hg	0.00041	18 -37	0.00042	18 – 37		
Mo	0.0028	18 – 37	0.00077	18 – 37	0.57	36 – 64
Ni	0.00016	18 – 37	0.000003	37 – 65	0.0072	16 – 36
Pb	0.0022	18 – 37	0.0055	18 – 37	0.11	36 – 64
Sb	0.011	18 – 37	0.0039	18 – 37	0.14	36 – 64
Se	0.012	37 – 65	0.0018	37 – 65		
Zn	0.011	37 – 65	0.029	37 – 65	0.093	36 – 64
Chloride	250	37 – 65			2000	36 – 64
Fluoride						
Sulphate	174	37 – 65	4	37 - 65	118	36 – 64
DOC						

7.3.2 Results of scenario calculations

The values of C_{eq} and k found in tables 7.2 and 7.3 have been used together with various values of the landfill height (H); the geometric surface area per unit volume (A) and the rate of infiltration into or percolation through the landfill (Inf) to calculate the concentration of C from

equation (7.1), and subsequently to determine whether C/C_{eq} is close to 1. A value of C/C_{eq} close to 1, where C is the concentration of a given component in the leachate at the bottom of the landfill, is an indication that the assumption, that the leaching of that component at the bottom of the landfill occurs under equilibrium-like conditions, is fulfilled.

Since the landfill height used in the TAC scenario calculations for landfills for non-hazardous and hazardous waste was 20 m and the corresponding rate of infiltration was 200 mm/year, it would seem reasonable to use $H = 20$ m and $Inf = 200$ mm/year as starting or “default” values. The value of A is more uncertain, but $A = 12$ m²/m³ corresponding to the waste being placed as 0.5 m by 0.5 m by 0.5 m cubes could be used as the starting point. The acceptance criteria for landfilling of monolithic waste being implemented in the UK are based on this assumption.

The results of the calculations of C/C_{eq} for various combinations of H , A and Inf for each of the three monolithic waste materials are presented in tables 7.4, 7.5 and 7.6.

Table 7.6 Results of scenario calculations for stabilised waste BCR-2. C/Ceq \approx 1 indicates equilibrium.

BCR-2 Test of equilibrium condition using Ceq and k from leaching tests								
Scenario:	A	B	C	D	E	F	G	H
H (m):	20	20	20	20	20	10	10	10
A (m ² /m ³):	12	4	4	4	4	12	4	1
Inf (mm/year):	200	300	100	50	10	200	50	10
Component	C /Ceq	C /Ceq	C /Ceq	C /Ceq	C /Ceq	C /Ceq	C /Ceq	C /Ceq
As	1.000	1.000	1.000	1.000	1.000	1.000	1.000	1.000
Ba	0.359	0.094	0.257	0.448	0.949	0.200	0.257	0.310
Cd	1.000	1.000	1.000	1.000	1.000	1.000	1.000	1.000
Cr	1.000	1.000	1.000	1.000	1.000	1.000	1.000	1.000
Cu	1.000	1.000	1.000	1.000	1.000	1.000	1.000	1.000
Hg	1.000	1.000	1.000	1.000	1.000	1.000	1.000	1.000
Mo	1.000	0.9874	1.0000	1.0000	1.0000	0.9999	1.0000	1.000
Ni	1.000	0.997	1.000	1.000	1.000	1.000	1.000	1.000
Pb	1.000	1.000	1.000	1.000	1.000	1.000	1.000	1.000
Sb	1.000	1.000	1.000	1.000	1.000	1.000	1.000	1.000
Se	0.999	0.797	0.992	1.000	1.000	0.972	0.992	0.997
Zn	1.000	1.000	1.000	1.000	1.000	1.000	1.000	1.000
Chloride	1.000	0.972	1.000	1.000	1.000	1.000	1.000	1.000
Fluoride								
Sulphate								
DOC								

From table 7.4 it can be seen that the equilibrium condition appears to be fulfilled or nearly fulfilled for all of the components in almost all of the scenarios for **StabW**. Ni seems to be the component which is closest to release rate control. However, Ni release only shows a significant deviation from equilibrium controlled conditions in scenario B, where the geometric surface area per unit volume (A) has been reduced to 1/3 and the rate of infiltration (Inf) has been increased by 50 % relative to the default scenario A. With the exception of fluoride and DOC, the equilibrium evaluation for StabW comprises all the components for which leaching limit values have been established in Council Decision 2003/33/EC.

For **BCR-1** data are not available for As, Hg, Se, chloride, fluoride and DOC. As table 7.5 shows, the equilibrium assumption is not fulfilled at all for Ni and Cd, and only partially fulfilled for Mo and sulphate. Smaller deviations from equilibrium conditions are also seen for Pb and Zn for some of the scenarios. Under the conditions upon which the scenarios in table 7.5 are based, the concentration of several components thus appears to be release controlled rather than equilibrium controlled. This is probably due to the high retention efficiency of the stabilised matrix, which contains less than 10 % of actual waste. It is therefore hardly typical of waste that destined for landfilling. The material is highly alkaline, and pH is not likely to be lowered significantly at the bottom of the landfill for a long period of time. If the same scenario calculations are performed using values of C_{eq} that are determined at pH = 10 for all com-

ponents, only the release of Ni and sulphate (and in some scenarios Mo) will show significant deviations from equilibrium behaviour.

For **BCR-2** data are not available for As, Hg, Se, chloride, fluoride and DOC. Table 7.6 shows that the equilibrium assumption is fulfilled for all the components analysed, except Cd in all scenarios and Zn in scenario B, where the geometric surface area per unit volume (A) has been reduced to $1/3$ and the rate of infiltration (Inf) has been increased by 50 % relative to the default scenario A. BCR-2 is also a highly alkaline waste material, and if the C_{eq} determined at $pH = 10$ l/kg is used in the calculations, the equilibrium assumption is fulfilled for all scenarios for all the parameters shown in table 7.6.

In Appendix 6 C_{eq} and C are shown as a function of the rate of infiltration, Inf , for $H = 10$ m and $H = 20$ m and $A = 20$ m²/m³, 12 m²/m³, 4 m²/m³ and 1 m²/m³ for each of the components shown in tables 7.4, 7.5 and 7.6 for StabW, BCR-1 and BCR-2. The figures in Appendix 6 provide a more detailed description of the dependency of the fulfilment of the equilibrium assumption on the rate of infiltration and the geometric surface area per unit volume than tables 7.4 to 7.6.

7.3.3 Conclusion concerning the equilibrium situation

Based on the scenario calculations presented in the previous section it would seem reasonable to assume that for most monolithic waste materials which are not excessively stabilised and which are landfilled under conditions similar to those used in the TAC calculations ($H = 20$ m and $Inf = 200$ mm/year), equilibrium conditions will control the release of most components at the bottom of the landfill. This may not be true for some components with limited interaction with other components (e.g. chloride and sodium). The concentration of those components may, to a large extent, be release rate/diffusion controlled. However, for some stabilised waste materials such as MSWI APC residues aimed for landfilling, the release of these components under release rate control is fairly extensive during the first period of exposure, which may lead to relatively high concentrations in the leachate over a longer period of time.

It should be noted that the model and the scenarios used are simplified and that a constant release rate is used instead of a rate that decreases with time. The parameters C_{eq} and k used in the scenario calculations have been derived from experiments on three monolithic waste materials, where only two (StabW and BCR-2) can be expected to be representative of waste to be landfilled. If diffusion is a limiting process for the release rate, the parameter k is dependent on the size of the monoliths and results from laboratory tank leaching tests on monolithic specimens will overestimate the value of k under field conditions. No actual information is available on the size of the geometric surface area per unit volume, A , so it has been estimated without any practical verification. Efforts should be

made to improve the model and the supporting data to enable a future review of these conclusions.

7.4 The source term under equilibrium-like conditions

7.4.1 Principle

If it is assumed that equilibrium conditions prevail at the bottom of the landfill, the source term in the step-wise impact assessment used to set acceptance criteria based on leaching (see chapter 5) may be based on the same principles as those used by the TAC to set the criteria for granular waste in Council Decision 2003/33/EC. Similarly, the test methods used to characterise the waste and check whether it complies with the criteria, should then be the same equilibrium-based leaching tests as those prescribed by the Council Decision for granular waste, i.e. column and/or batch leaching tests (CEN/TS 14405 and EN 12457 – parts 1 to 3) carried out on material crushed to < 4 mm. This means that in the test result there will be little or no benefit from the mechanical effect of stabilization, reflecting the fact that for most constituents the mechanical stabilization does not appear to have a significant influence on the resulting concentrations in the leachate at the bottom of the site within the timeframes considered. Any chemical effect of the stabilization that influences the composition of the leachate under landfill conditions will, of course, also have an influence on the test results and will therefore be taken into account.

The source term will consist of a description of the estimated or assumed flow or percolation of water/leachate through the landfill and a description of the estimated (equilibrium) concentration of the leachate as a function of L/S. In order to describe the source, the physical landfill scenario must also be described. The scenarios and conditions described below are those used in the TAC calculations.

7.4.2 The physical landfill scenario

The physical scenarios used in the TAC calculations to derive granular waste acceptance criteria for non-hazardous waste landfills (non-hazardous waste landfills receiving stable, non-reactive hazardous waste) and hazardous waste landfills are described in table 7.7. The only difference between the two scenarios is that for non-hazardous waste landfills the clay part of the bottom liner (or the geological barrier) is 1 m thick, whereas it is 5 m thick for the hazardous waste landfill.

Table 7.7 Description of the physical scenarios for landfills for non-hazardous and hazardous waste used in the TAC calculations of limit values for granular waste (DHI & ECN 2003, DHI & MST 2005).

Parameter	Unit	Landfill for non-haz waste	Landfill for haz waste
Landfill height	m	20	20
Landfill length	m	200	200
Landfill width	m	200	200
Surface area	m ²	40,000	40,000
Volume	m ³	800,000	800,000
Waste porosity	-	0.3	0.3
Dry density of waste	t/m ³	1.5	1.5
Dry weight of landfilled waste	t	1,200,000	1,200,000
Permeability of waste	m/s	1 x 10 ⁻⁵	1 x 10 ⁻⁵
Hydraulic conductivity of top cover	mm/year	Variable (31,5 – 200)	Variable (31,5 – 200)
Type of bottom liner	-	Composite	Composite
Thickness of bottom liner	m	1	5
Permeability of bottom liner	m/s	10 ⁻⁹	10 ⁻⁹

7.4.3 Hydraulic conditions

The hydraulic conditions applied in the TAC scenarios were the same for landfills for non-hazardous waste receiving stable, non-reactive hazardous waste and landfills for hazardous waste. Table 7.8 shows the rate of infiltration through the top cover, the rate of percolation of leachate through the (artificial) bottom liner into the unsaturated zone (the clay barrier) and the annual amount of leachate that must be collected in the drainage system for subsequent treatment and discharge. The scenario is obviously simplified and does e.g. not take any water storage capacity of the waste into account. Furthermore, the subsequent calculations were carried out as if the entire amount of waste was present from the start. The simplifications are justified by the fact that the calculations were supposed to be general and not reflect any specific filling pattern or physical waste properties (e.g. initial water content and retention capacity).

Table 7.8 Description of the assumed water balances used in the TAC calculations of limit values for acceptance of granular waste at landfills for non-hazardous waste (receiving stable, non-reactive hazardous waste) and hazardous waste (DHI & ECN 2003, DHI & MST 2005).

Period Years	Infiltration through the top/top cover mm/year	Infiltration through the bottom liner mm/year	Leachate, which must be collected and trea- ted/discharged mm/year
0 – 30	200	Increases from 2 to 31.5	Decreases from 198 to 168.5
30 – 60	0	0	0
60 – 80	Increases gradually from 0 to 200	Increases gradually from 0 to 31.5	
80 – 110		31.5	Increases gradually from 0 to 168.5
110 - ∞	200	31.5	168.5
Comments:	No top cover during the operation period, installa- tion of artificial liner at the time of closure.	Composite liner: The clay liner remains effective, whereas the artificial liner totally degrades over a period of 200 years.	Note: The calculations are based on the assumption of “eternal” collection of leachate beyond the 110 years.

The description is divided into a number of time periods according to the changes in conditions occurring at the top cover and/or the bottom liner:

During the first 30 years (the operation period), the waste is uncovered and an annual infiltration of 200 mm is assumed. The initial efficiency of the artificial bottom liner is assumed to be 99 % corresponding to an initial leakage rate of 2 mm/year. It is assumed that the artificial liner will deteriorate totally at an almost linear rate over the period 0 to 200 years. The potential leakage through the bottom liner will reach the assumed leakage capacity of the underlying clay liner (31.5 mm/year) at the end of the operation period. The amount of leachate to be collected and treated will decrease from 198 mm/year to 168.5 mm/year during this period.

At the end of the operation period the landfill will be covered by a top cover including an artificial top liner, which is assumed to be 100 % effective for 30 years (the period for year 30 to year 60). During this period, no leachate infiltrates and hence no leachate leaks through the bottom liner and no leachate is collected for treatment and discharge.

After 30 years, i.e. from year 60, the artificial top liner starts to lose its functionality, and over the next 50 years it deteriorates totally, causing the a gradual increase of the annual infiltration through the top cover from 0 mm to 200 mm during the period from year 60 to year 110. During the first 20 years of that period, i.e. from year 60 to year 80, the annual discharge through the bottom liner increases gradually from 0 mm to 31.5 mm, corresponding to the permeability of the clay liner assuming a gradient of 1 m/m. During the same period, the amount of leachate to be

collected and treated/discharged will remain 0 mm/year. During the period 80 to 110 years the rate of infiltration increases from 31.5 mm/year to 200 mm/year, the leakage through the bottom liner remains at 31.5 mm/year, and the amount of leachate to be collected and treated increases from 0 to 168.5 mm/year.

From 110 years and onwards, the annual rate of infiltration through the top cover remains at 200 mm, the leakage through the clay bottom liner remains at 31.5 mm/year, and the annual amount of leachate to be collected is 168.5 mm. It should be noted that the scenario assumes that continued collection (and treatment) of leachate beyond year 110 is possible and that it will, in principle, be carried out eternally.

7.4.4 Leachate composition as a function of L/S

In the equilibrium situation the description of the leachate composition as a function of L/S for the purposes of modelling is based on several simplifying assumptions, including the following:

1. All waste types and all units in a landfill produces identical leachate as far as the modelling is concerned (i.e. the waste is considered the same at all locations within the landfill), and the influence of waste-waste interactions on the leachate quality is neglected. When a waste type subsequently is tested in relation to the criteria, the results are evaluated as if the entire landfill consisted of this type of waste.
2. The landfill behaves in the same way as the waste a large column or a lysimeter test, and the flux through the bottom of the landfill (or into the leachate collection system) may be estimated from laboratory column tests or lysimeter tests and information on the hydraulic conditions.
3. The leaching of the inorganic components and a few organic components from largely mineral or inorganic waste may be described as an exponentially decreasing function of L/S or time, based on a continuously stirred tank reactor (CSTR) model (see e.g. Hjelm et al., 2001).

Using the CSTR model the concentration of a particular component in the leachate/eluate is described by the following equation:

$$C = C_0 e^{-(L/S) \kappa} \quad (7.2)$$

where

C is the concentration of the component in the leachate as a function of L/S (mg/l),

C_0 is the initial peak concentration of the component in the leachate (mg/l),

L/S is the accumulated liquid to solid ratio corresponding to the concentration C (l/kg),

κ is a kinetic constant describing the rate of decrease of the concentration as 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.

By integrating equation (7.2) with respect to L/S , the amount, E (in mg/kg) of the component in question released over the period of time it takes for L/S to increase from 0 to the value corresponding to C can be calculated:

$$E = (C_0/\kappa) (1 - e^{-(L/S)\kappa}) \quad (7.3)$$

Figure 7.4 illustrates the influence of the parameter κ on the release rate. Larger values of κ lead to a faster decrease of the concentration of a given component in the leachate as a function of L/S . For $\kappa = 5$ kg/l, C has decreased to less than 1 % of C_0 at $L/S = 1$ l/kg, whereas for $\kappa = 0.01$ kg/l, C has decreased only to 90 % of C_0 at $L/S = 10$ l/kg. In principle, κ depends both on the waste material and the component in question. However, only limited data have been available for the calculation of κ values, and therefore the same κ was used for all waste types in the TAC calculations. κ values for inorganic components determined by Aalbers et al. (1996) for secondary construction materials were used in the TAC calculations. These values, which are shown in table 7.9, will also be used in this context.

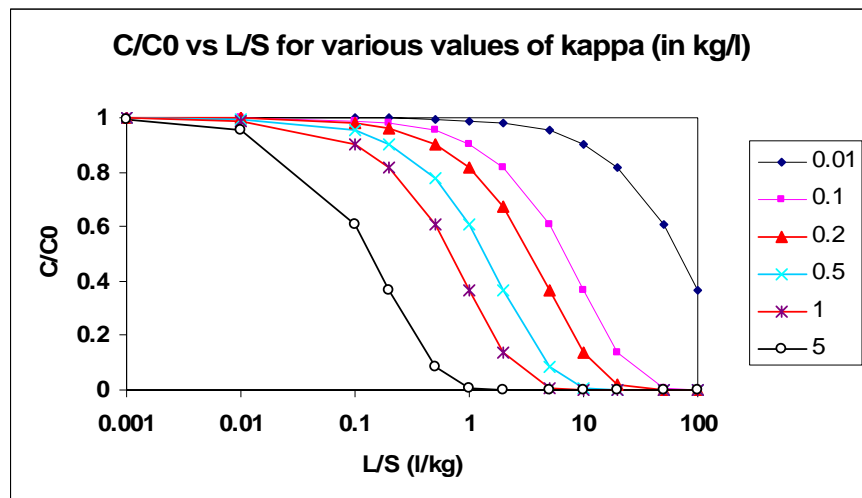


Figure 7.4 C/C_0 as a function of L/S based on equation 7.2 for various values of κ .

Table 7.9 Average values and 95 % confidence intervals for κ for a number of components (Albers et al. 1996). The value for DOC has been determined by ECN (DHI and ECN, 2003).

Component	Average values and 95 % confidence intervals for κ (kg/l)	Number of determinations, n
As	0,03 ± 0,05	44
Ba	0,15 ± 0,04	55
Cd	0,50 ± 0,10	37
Cr	0,18 ± 0,03	82
Cu	0,28 ± 0,03	90
Hg	0,05 ± 0,03	5
Mo	0,35 ± 0,04	76
Ni	0,29 ± 0,05	37
Pb	0,27 ± 0,06	52
Sb	0,11 ± 0,07	33
Se	0,38 ± 0,18	10
Zn	0,28 ± 0,05	41
Klorid	0,57 ± 0,07	45
Fluorid	0,22 ± 0,14	6
Sulfat	0,33 ± 0,05	49
DOC	0.17	

It should be kept in mind, though, that the values presented in table 7.9 have been determined from leaching data on waste with a relatively low leaching potential, and that they do not provide a representative description of the leaching curve for all waste types. This is due partly to the oversimplification in using the CSTR model, partly to the fact that the κ values determined in reality are material-specific. There is a need for development of better and more differentiated κ values or a more sophisticated description of the release for future use.

7.4.5 Calculation of the flux of contaminants as a function of time

The relationship between time and L/S may be derived from the following equation (Hjelmar, 1990):

$$t = (L/S) d H/I \quad \text{where} \quad (7.4)$$

t is the time since the landfill started producing leachate (years)

L is the accumulated volume of leachate produced at time t (m³)

S is the total dry mass of waste deposited at the landfill (tonnes)

d is the average dry bulk density of the deposited waste (tonnes/m³)

H is the average height of the landfill (m)

I = the rate of infiltration into and percolation through the landfill (m/year)

By substituting t for L/S in equations (7.2) and (7.3) by means of equation (7.4) and using the values of d and H from table 7.7 and letting I vary with time as described in table 7.8, the release of a given contaminant (characterised by a κ value in table 7.9) may be described as a function of time in terms of (C/C_0) . For the purpose of the transport modelling, the concentration of a given component is normalised and described as C/C_0 for that component. The annual amount of leachate produced may be calculated from the information in table 7.7 and 7.8, and the normalised flux can also be calculated.

8. Acceptance criteria for the equilibrium situation

8.1 Criteria corresponding to the values in Council Decision 2003/33/EC

In chapter 7 it was shown that it seems reasonable to assume that the release of contaminants at the bottom of the landfill will be controlled by equilibrium/solubility for most components and many “monolithic” waste materials under expected “normal” conditions. This would correspond to the conditions assumed in the TAC calculations of acceptance criteria for granular waste materials, and hence there would be no reason to change the methodology of the calculations used by the TAC for the purpose of establishing criteria for “monolithic” waste. The water balance of a “monolithic” landfill may, however, be different from that of a landfill for granular waste, either because of requirements or restrictions or because of differences in landfill design/operation and/or waste properties.

Table 8.1 (C_0 or $L/S = 0 - 0.1$ l/kg in CEN/TS 14405), table 8.2 ($L/S = 2$ l/kg) and table 8.3 (10 l/kg) show the acceptance criteria for non-hazardous and hazardous waste landfills that would result from a calculation corresponding to that performed for granular waste by the TAC as described in chapter 5, Appendix 5 and section 7.4, using the same water balance assumptions as for the granular waste (table 7.8). The tables also show the granular waste acceptance criteria that were actually adopted after discussion in the TAC of the modelling results.

From tables 8.1, 8.2 and 8.3 it can be seen that for most parameters, the actual waste acceptance criteria (WAC) have been set somewhat higher than the levels resulting from the model calculations. The only exceptions are Cr (both for landfills for non-hazardous and hazardous waste) and Se (for landfills for non-hazardous waste) for which the criteria has been set lower than the values resulting from the model calculations. Table 8.4 shows the ratios between the actual and the calculated WAC.

Table 8.1 Waste acceptance criteria (WAC) calculated for landfilling of granular waste using the methodology and conditions applied by the TAC as well as the actually adopted WAC that are listed in Council Decision 2003/33/EC. The WAC shown here corresponds to C₀.

Parameter	Landfill for non-hazardous waste		Landfill for hazardous waste	
	Calculated WAC	Actual WAC	Calculated WAC	Actual WAC
	mg/l	mg/l	mg/l	mg/l
As	0.21	0.3	0.39	3
Ba	9.7	20	12	60
Cd	0.29	0.3	0.78	1.7
Cr	5.9	2.5	17	15
Cu	1.8	30	4.2	60
Hg	0.012	0.03	0.013	0.3
Mo	2.3	3.5	5.3	10
Ni	1.9	3	5.5	12
Pb	0.91	3	2.6	15
Sb	0.076	0.15	0.11	1
Se	0.24	0.2	0.47	3
Zn	6.2	15	16	60
Chloride	3330	8500	3450	15000
Fluoride	22	40	30	120
Sulphate	3150	7000	3180	17000
DOC	122	250	129	320

Table 8.2 Waste acceptance criteria (WAC) calculated for landfilling of granular waste using the methodology and conditions applied by the TAC as well as the actually adopted WAC that are listed in Council Decision 2003/33/EC. The WAC shown here corresponds to L/S = 2 l/kg.

Parameter	Landfill for non-hazardous waste		Landfill for hazardous waste	
	Calculated WAC	Actual WAC	Calculated WAC	Actual WAC
	mg/kg	mg/kg	mg/kg	mg/kg
As	0.4	0.4	0.75	6
Ba	17	30	21	100
Cd	0.36	0.6	0.98	3
Cr	9.9	4	28	25
Cu	2.7	25	6.4	50
Hg	0.023	0.05	0.024	0.5
Mo	3.3	5	7.6	20
Ni	3.0	5	8.3	20
Pb	1.4	5	4.0	25
Sb	0.14	0.2	0.19	2
Se	0.33	0.3	0.66	4
Zn	9.4	25	25	90
Chloride	3970	10000	4120	17000
Fluoride	36	60	49	200
Sulphate	4610	10000	4650	25000
DOC	206	380	220	480

Table 8.3 Waste acceptance criteria (WAC) calculated for landfilling of granular waste using the methodology and conditions applied by the TAC as well as the actually adopted WAC that are listed in Council Decision 2003/33/EC. The WAC shown here corresponds to L/S = 10 l/kg.

Parameter	Landfill for non-hazardous waste		Landfill for hazardous waste	
	Calculated WAC	Actual WAC	Calculated WAC	Actual WAC
	mg/kg	mg/kg	mg/kg	mg/kg
As	1.8	2	3.3	25
Ba	50	100	58	300
Cd	0.57	1	1.5	5
Cr	27	10	78	70
Cu	5.9	50	14	100
Hg	0.095	0.2	0.1	2
Mo	6.4	10	15	30
Ni	6.3	10	18	40
Pb	3.2	10	8.8	50
Sb	0.46	0.7	0.66	5
Se	0.62	0.5	1.2	7
Zn	21	50	55	200
Chloride	5830	15000	6040	25000
Fluoride	90	150	120	500
Sulphate	9230	20000	9290	50000
DOC	590	800	620	1000

Table 8.4 Ratios between the actual and calculated WAC as presented in table 8.1, 8.2 and 8.3.

Para-meter	Landfill for non-hazardous waste			Landfill for hazardous waste		
	Actual WAC/Calculated WAC			Actual WAC/Calculated WAC		
	C ₀	L/S = 2 l/kg	L/S = 10 l/kg	C ₀	L/S = 2 l/kg	L/S = 10 l/kg
As	1.5	1.0	1.1	7.8	8.0	7.6
Ba	2.1	1.8	2.0	5.1	4.8	5.2
Cd	1.0	1.7	1.8	2.2	3.1	3.3
Cr	0.43	0.40	0.37	0.88	0.89	0.90
Cu	17	9.3	8.5	14	7.8	7.1
Hg	2.5	2.2	2.1	24	21	20
Mo	1.5	1.5	1.6	1.9	2.6	2.0
Ni	1.5	1.7	1.6	2.2	2.4	2.2
Pb	3.3	3.6	3.1	5.8	6.3	5.7
Sb	2.0	1.4	1.5	9.3	11	7.6
Se	0.84	0.91	0.81	6.4	6.1	5.8
Zn	2.4	2.7	2.4	3.7	3.6	3.6
Chloride	2.6	2.5	2.6	4.3	4.1	4.1
Fluoride	1.8	1.7	1.7	4.0	4.1	4.2
Sulphate	2.2	2.2	2.2	5.3	5.4	5.4
DOC	2.0	1.8	1.4	2.5	2.2	1.6

In cases where the WAC from Council Decision 2003/33/EC have been accepted for granular waste, and no major differences between the water balances for landfills for granular and monolithic waste, respectively, are expected or have been prescribed, there would be no reason to choose acceptance criteria for monolithic waste that are different from those for granular waste. The actual WAC shown in table 8.1 for C₀, in table 8.2 for testing at L/S = 2 l/kg and in table 8.3 for testing at L/S = 10 l/kg would therefore apply.

8.2 Criteria corresponding to other conditions than those used by the TAC

8.2.1 Changes in the water balance and other parameters

The same methodology as that described above may be used to calculate new criteria for acceptance of waste at landfills if the scenarios and the model parameters are changed, e.g. to provide a better fit to local or national environmental conditions and landfill strategy and practice. Examples of conditions and parameters that could be subject to changes are: landfill dimensions (length, width, height), waste density, the general rate of infiltration, the water balance for the landfill, κ and K_d values for the contaminants, groundwater quality criteria at the POC, location of the POC and the background concentration of the contaminants in the

groundwater (assumed = 0 in the TAC calculations). It should be noted that for granular waste this cannot (for legal reasons) lead to national criteria that are more lenient than those currently listed in Council Decision 2003/33/EC. It is a question whether this would also be the case for a “monolithic” waste landfill with leaching conditions based on equilibrium/solubility control.

For the purpose of illustration, acceptance criteria have been calculated for a water balance description adjusted to fit the requirements implemented in Sweden (Miljö- och samhällsbyggnadsdepartementet, 2001): The rate of infiltration through the top liner must not exceed 50 mm/year for non-hazardous waste landfills and 5 mm/year for hazardous waste landfills after establishment of the top cover. The bottom liner must not allow a leakage exceeding 50 mm/year for non-hazardous waste landfills and 5 mm/year for hazardous waste landfills. The calculations have been carried out in chapter 5 and section 7.4 using the same conditions as those used in the TAC calculations except for the water balances, which are shown in table 8.5 and table 8.6.

Table 8.5 Description of the assumed water balances corresponding to the requirements implemented in Sweden for landfills for non-hazardous waste.

Period Years	Infiltration through the top/top cover mm/year	Infiltration through the bottom liner mm/year	Leachate, which must be collected and treated/discharged mm/year
0 – 30	200	50	150 mm
30 – ∞	50	50	0
Comments	No top cover during the operation period, installation of infiltration-reducing top cover at the time of closure (maximum leakage: 50 mm/year).	The bottom liner must not allow annual leakage of leachate that exceeds 50 mm.	Leachate collection ceases when leakage control shifts from bottom liner to top cover

Table 8.5 Description of the assumed water balances corresponding to the requirements implemented in Sweden for landfills for hazardous waste.

Period Years	Infiltration through the top/top cover mm/year	Infiltration through the bottom liner mm/year	Leachate, which must be collected and treated/discharged mm/year
0 – 30	200	5	195 mm
30 – ∞	5	5	0
Comments	No top cover during the operation period, installation of infiltration-reducing top cover at the time of closure (maximum leakage: 5 mm/year).	The bottom liner must not allow an annual leakage of leachate that exceeds 5 mm.	Leachate collection ceases when leakage control shifts from bottom liner to top cover

The results are presented in table 8.6 for non-hazardous waste landfills and in table 8.7 for hazardous waste landfills in terms of criteria for C_0 and for leached amount at $L/S = 10$ l/kg. The tables also show some of the parameters used (K_d , κ , groundwater quality criteria) as well as the calculated attenuation factors and the time needed for the groundwater peak concentration to occur. For the purpose of comparison, the calculated and actual limit values produced by the TAC is also shown for leaching at $L/S = 10$ l/kg. All results refer to POC 1 (20 m downstream of the landfill).

Table 8.6 Waste acceptance criteria (WAC) for a landfill for non-hazardous waste calculated for granular (and monolithic) waste using the methodology and conditions applied by the TAC but adjusted to the Swedish water balance requirements (see table 8.4). WAC calculated and actually set by the TAC are also shown. C_{GWC} = ground water quality criteria and f_a = attenuation factor (see sections 5.8 and 5.9). All results refer to POC 1 (20 m downstream).

Component	Kd	Kappa	C_{GWC}	f_a	Time of peak	WAC in this case		EU TAC ($L/S = 10$ l/kg)	
						C_0	$L/S = 10$ l/kg)	WAC (calculated)	WAC (actual)
	l/kg	kg/l	mg/l		Years	mg/l	mg/kg	mg/kg	mg/kg
As	50	0.03	0.01	8	7428	0.08	0.69	1.8	2
Ba	2	0.15	0.7	7	425	4.9	25	50	100
Cd	20	0.5	0.004	12	1990	0.048	0.10	0.57	1
Cr	100	0.18	0.05	15	8550	0.75	3.5	27	10
Cu	14	0.28	0.05	9	1731	0.45	1.5	5.9	50
Hg	1	0.05	0.001	7	294	0.007	0.055	0.095	0.2
Mo	10	0.35	0.07	9	1274	0.63	1.7	6.4	10
Ni	50	0.29	0.02	14	4521	0.28	0.91	6.3	10
Pb	50	0.27	0.01	14	4602	0.14	0.48	3.2	10
Sb	5	0.11	0.005	8	917	0.04	0.24	0.46	0.7
Se	5	0.38	0.01	9	734	0.09	0.23	0.62	0.5
Zn	30	0.28	0.1	12	3102	1.2	4.0	21	50
Chloride	0	0.57	250	7	42	1750	3060	5830	15000
Fluoride	2	0.22	1.5	8	401	12	49	90	150
Sulphate	0	0.33	250	7	48	1750	5107	9230	20000
DOC	0	0.17	10	7	54	70	337	590	800

Table 8.7 Waste acceptance criteria (WAC) for a landfill for hazardous waste calculated for granular (and monolithic) waste using the methodology and conditions applied by the TAC but adjusted to the Swedish water balance requirements (see table 8.5). WAC calculated and actually set by the TAC are also shown. C_{GWC} = ground water quality criteria and f_a = attenuation factor (see sections 5.8 and 5.9). All results refer to POC 1 (20 m downstream).

Component	Kd	Kappa	C_{GWC}	f_a	Time of peak	WAC in this case		EU TAC (L/S = 10 l/kg)	
						C_0	L/S =10 l/kg	WAC (calculated)	WAC (actual)
	l/kg	kg/l	mg/l		Years	mg/l	mg/kg	mg/kg	mg/kg
As	50	0.03	0.01	71	51705	0.71	6.1	1.8	25
Ba	2	0.15	0.7	66	3136	46	239	50	300
Cd	20	0.5	0.004	103	11892	0.41	0.82	0.57	5
Cr	100	0.18	0.05	120	47962	6	28	27	70
Cu	14	0.28	0.05	82	11231	4.1	14	5.9	100
Hg	1	0.05	0.001	64	2271	0.064	0.50	0.095	2
Mo	10	0.35	0.07	81	8322	5.7	16	6.4	30
Ni	50	0.29	0.02	112	25984	2.2	7.3	6.3	40
Pb	50	0.27	0.01	110	26660	1.1	3.8	3.2	50
Sb	5	0.11	0.005	67	6640	0.34	2.0	0.46	5
Se	5	0.38	0.01	75	4987	0.75	1.9	0.62	7
Zn	30	0.28	0.1	96	18867	9.6	32	21	200
Chloride	0	0.57	250	67	347	16750	29288	5830	25000
Fluoride	2	0.22	1.5	67	2911	101	406	90	500
Sulphate	0	0.33	250	65	378	16250	47426	9230	50000
DOC	0	0.17	10	64	414	640	3077	590	1000

Table 8.7 Waste acceptance criteria (WAC) for a landfill for hazardous waste calculated for granular (and monolithic) waste using the methodology and conditions applied by the TAC but adjusted to the Swedish water balance requirements (see table 8.5). WAC calculated and actually set by the TAC are also shown. C_{GWC} = ground water quality criteria and f_a = attenuation factor (see sections 5.8 and 5.9). All results refer to POC 1 (20 m downstream).

Component	Kd	Kappa	C_{GWC}	f_a	Time of peak	WAC in this case		EU TAC (L/S = 10 l/kg)	
						C_0	L/S =10 l/kg	WAC (calculated)	WAC (actual)
	l/kg	kg/l	mg/l		Years	mg/l	mg/kg	mg/kg	mg/kg
As	50	0.03	0.01	71	51705	0.71	6.1	1.8	25
Ba	2	0.15	0.7	66	3136	46	239	50	300
Cd	20	0.5	0.004	103	11892	0.41	0.82	0.57	5
Cr	100	0.18	0.05	120	47962	6	28	27	70
Cu	14	0.28	0.05	82	11231	4.1	14	5.9	100
Hg	1	0.05	0.001	64	2271	0.064	0.50	0.095	2
Mo	10	0.35	0.07	81	8322	5.7	16	6.4	30
Ni	50	0.29	0.02	112	25984	2.2	7.3	6.3	40
Pb	50	0.27	0.01	110	26660	1.1	3.8	3.2	50
Sb	5	0.11	0.005	67	6640	0.34	2.0	0.46	5
Se	5	0.38	0.01	75	4987	0.75	1.9	0.62	7
Zn	30	0.28	0.1	96	18867	9.6	32	21	200
Chloride	0	0.57	250	67	347	16750	29288	5830	25000
Fluoride	2	0.22	1.5	67	2911	101	406	90	500
Sulphate	0	0.33	250	65	378	16250	47426	9230	50000
DOC	0	0.17	10	64	414	640	3077	590	1000

The results in table 8.6 show that the criteria for acceptance of waste at landfills for non-hazardous waste calculated for the Swedish situation generally are somewhat more stringent than those calculated by the TAC. This is due to the fact that the Swedish conditions allow a leakage of 50

mm/year from non-hazardous waste landfills, whereas on 31.5 mm/year were allowed in the TAC calculations.

The results in table 8.7 show that criteria for acceptance of waste at landfills for hazardous waste calculated for the Swedish situation generally are less stringent than those calculated by the TAC. This is due to the fact the Swedish conditions only allow a leakage of 5 mm/year from hazardous waste landfills, which is less than the 31.5 mm/year allowed in the TAC calculations. However, only the criteria calculated for chloride and DOC exceed the actual waste acceptance criteria set by the TAC.

8.2.2 Criteria related to surface water quality

The TAC calculations that led to the leaching criteria in Council Decision 2003/33/EC was based exclusively on an evaluation of the risk to downstream groundwater quality (although many values were adjusted during the subsequent negotiations). In some of the Nordic countries or in some areas of the Nordic countries, the risk to surface water bodies is of more concern than the risk to groundwater. The water bodies of concern may be rivers, brooks, lakes or marine waters. In view of the requirements of the Landfill Directive in terms of geological barriers it seems reasonable to assume that a landfill will never be placed closer than 20 m to a surface water body. In principle, the impact on the surface water body could be evaluated by coupling a fourth model to the series of three models shown in figure 5.1. The fourth model should describe the transport/dilution of the contaminants in the surface water body. The input to the model would be the flow of groundwater and the flux of contaminants at the point where the leachate plume enters into the surface water body. A new POC would have to be chosen, either close to the point of entry into the surface water body (taking only initial dilution into account) or at some distance, taking more of the dilution potential of the surface water body into account. This could be done on a site-specific basis, but that would not support the development of general criteria. In some countries, general quality criteria exists for surface water bodies, and they (or some criteria developed for this specific purpose) could be applied at a chosen POC. The coupled models could then be run forwards as described for the three models in Chapter 5, and once ratio between the highest concentration in the leachate at the base of the landfill and the peak concentration at the POC (the attenuation factor) has been determined, the criteria may be calculated the same way as in the groundwater protection case. In this way it would be possible to develop criteria that would ensure adequate protection of surface water.

In some cases it is possible to get a first impression of the degree of protection of the surface water quality provided by the existing leaching criteria. If it is assumed that the downstream boundary of the landfill is placed 20 m from the surface water body in question, then the groundwa-

ter quality at that point (the TAC POC) may be calculated from the C_0 in the criteria and attenuation factor found by the modelling (this is necessary because most of the calculated criteria were changed during the negotiation phase in the TAC before they became actual WAC). That groundwater quality may then be divided by an appropriated dilution factor (for example, an initial dilution of 10 times is often used as a rule of thumb for marine coastal waters in Denmark) and compared with the water quality criteria for the surface water body.

9. WAC for landfilling of monolithic waste in other countries

Most of the EU member states are still in the process of implementing criteria for landfilling of monolithic waste and actual WAC for monolithic waste are only available from a few countries. The following information has been obtained from the UK, the Netherlands and France, which have set or proposed WAC for landfilling of monolithic waste. The criteria set by the three countries differ substantially from each other and are resulting from different approaches.

9.1 Criteria proposed by the United Kingdom

The calculation of the criteria proposed by the UK has been described by Hall and Drury (2004). The calculations are based upon the assumption that the waste is placed in the landfill in monolithic cubes with 0.5 m sides corresponding to an available surface area (or geometric surface area per unit volume) of 12 m²/m³. It is further assumed that the release occurs under diffusion control through the surface and that the cumulative release can be described as a linear function of the square root of time:

$$\frac{(\text{Cum. release to time 2})}{(\sqrt{\text{time1}})} = \frac{(\sqrt{\text{time2}})}{(\sqrt{\text{time1}})} \quad (9.1)$$

In the case of diffusion control, the cumulative release measured during 64 days in the Dutch standard tank leaching test may be used to predict the cumulative emission at any future point in time. Using this as the source term, the UK model (an adjusted version of one of the models used in the TAC calculations of granular waste acceptance criteria) was run forwardly for each component in an iterative fashion to derive the maximum cumulative release from a 64 day test that would result in the appropriate compliance point being met at all times within the simulation period (up to 100,000 years). Equation (9.1) may also be used to convert the maximum cumulative release at 64 days to a shorter duration for the purpose of setting limiting cumulative release values for a shorter compliance test, see below.

Hall and Drury (2004) have investigated and/or commented upon the sensitivity of the modelling results to waste porosity, waste filling sequence, geometric surface area per unit volume (block size), solubility limitation and total availability/depletion. They found only limited effects of variations of the porosity between 0.005 and 0.02 (a value of 0.02 was used in the model), and limited effects of simulating a gradual filling of the landfill rather than assuming that the landfill was filled up from day 1 (as used in the modelling). Changes in block size influenced the results of different components differently, mainly depending on their release characteristics and availability. The influence of block size increased more rapidly as block sides decreased below 0.4 m. Solubility limitations were observed experimentally but not included in the modelling. The possible effect of availability limitation during the simulation period was acknowledged but not included in the modelling.

When setting the criteria for landfilling of monolithic waste, the values resulting from the model calculations have been modified using a multiplier derived from the ratio between the limit values calculated with the UK model and the actual values set by the TAC for granular waste (similar the ratios shown in table 8.3). The proposed UK acceptance criteria for landfilling of monolithic waste are as follows (UK Environmental Agency, 2004):

The monolithic waste to be deposited must be a block with sides of at least 0.4 m each. Characterization testing includes determination of leaching properties using the 64-day NEN 7345 tank leaching test for monolithic waste on samples > 40 mm in any direction. The criteria to be met in terms of accumulative release over 64 days (8 changes of water) are shown in table 9.1. For compliance purposes, a shorter (4 days) version of tank leaching test corresponding to the first 4 steps of NEN 7345 may be used, and the criteria to be met by the accumulative release over these four steps are $\frac{1}{4}$ of the values shown in table 9.1 (derived from equation (9.1)). The waste to be stabilised into a monolithic form prior to landfilling must comply with the granular WAC before it enters the monolithic treatment plant (table 9.2).

Table 9.1 Proposed leaching limit values for monolithic waste in the UK. The limit values correspond to the cumulative releases measured over 64 days using the Dutch tank leaching test (NEN 7345).

Parameter	Stable non-reactive hazardous waste in non-hazardous waste landfills	Hazardous waste in hazardous waste landfills
	mg/m ²	mg/m ²
As	1.3	20
Ba	45	150
Cd*	0.2 (0.03)	1.0 (0.04)
Cr	5	25
Cu	45	60
Hg*	0.1 (0.01)	0.4 (0.01)
Mo	7	20
Ni	6	15
Pb	6	20
Sb	0.3	2.5
Se	0.4	5
Zn	30	100
Chloride	10000	20000
Fluoride	60	200
Sulphate	10000	20000
DOC	must be determined	must be determined
pH	must be determined	must be determined
Conductivity	must be determined	must be determined

* No waste that exceeds the more lenient values for list 1 parameters (Cd and Hg) will be permitted in a landfill. An operator who wishes to take a waste that meets the more lenient values but fails to meet the more stringent ones will be required to conduct a risk assessment for his site in order to demonstrate that there will be no discernible discharge of these listed substances to groundwater (the same regulations apply to granular waste).

Table 9.2 Proposed criteria for wastes entering the monolithic treatment plant in the UK.

Parameter	Stable non-reactive hazardous waste in non-hazardous waste landfills	Hazardous waste in hazardous waste landfills
TOC	6 %	6 %
LOI	10 %	10 %

The UK leaching criteria for monolithic waste is based on the assumption that the release of contaminants is controlled by diffusion. However, it was shown in chapter 7 that the release at the bottom of the landfill in many cases for many components appears to be controlled by solubility/equilibrium. If this is the case, then the criteria proposed by the UK may not truly reflect the leaching conditions in the landfill and they may possibly be somewhat more stringent than necessary (provided the other assumptions are valid).

9.2 Criteria used by the Netherlands

In 2002, the Dutch Ministry of the Environment set criteria for landfilling of stabilised hazardous waste (VROM, 2002). The criteria specify that the stabilising agents must not constitute more than 25 % of the volume of the stabilised material. Criteria are set for the compressive strength of the monolithic material: After 28 days of hardening the compressive strength measured according to EN 12394 must exceed 0.1 N/mm^2 (= 0.1 MPa). The leaching properties must be measured by the tank leaching test NEN 7345 (using 8 changes of distilled water), and the cumulative release over 64 days must comply with the values listed in table 9.3. In addition, no more than 1 weight % of the test body used in the tank test must fall to the bottom of the tank during the test.

Table 9.3 Dutch leaching criteria for acceptance of hazardous monolithic waste for landfilling. The criteria corresponds to the cumulative release over 64 days measured by the Dutch tank leaching test NEN 7345.

Parameter	Cumulative release over 64 days (mg/m ²)
As	50
Ba	1500
Cd	5
Co	60
Cr	500
Cu	500
Hg	1
Mo	900
Ni	400
Pb	1000
Sb	50
Se	60
Sn	50
V	1500
W	250
Zn	800
Br	5000
Chloride	250000
CN (total)	220
CN (free)	20
Fluoride	2500
Sulphate	250000

The Dutch criteria for monolithic waste reflect tank leaching test results from waste materials believed to belong in such landfills, and they are not based on scenario calculations and risk assessments (van der Sloot, 2005). It is therefore difficult to evaluate the degree of environmental protection provided by these criteria.

9.3 Criteria implemented by France

The French implementation of the criteria for acceptance of monolithic waste at landfills for hazardous waste is based on the French (AFNOR) standard XP X 31-211. It is a one-step tank leaching test with a contact time of 24 hours at an L/S ratio of 10 l/kg. The eluent used is demineralised water. The required size and shape of the test specimen is a 4 cm by 8 cm cylinder (4 cm diameter and 8 cm length). The criteria to be observed are the same as those listed for granular waste for L/S = 10 l/kg in Council Decision 2003/33/EC (La ministre de l'écologie et du développement durable, 2003).

The test specimen has a surface area of 125.66 cm² and a volume of 100.53 cm³. For a material with a dry density of 1 kg/l, this corresponds to an L/A (liquid to surface ratio) of 8 cm³/cm². For a material with a dry density of 1.5 kg/l, it corresponds to L/A = 12 cm³/cm². The dry density is not addressed in the test, and the French implementation does not establish a connection between the expected leaching mechanism in the tank test and the landfill scenario. The amount of a given component

leached from a given monolithic material during the test will most likely be proportional to the surface area of the test body. However, the test data are interpreted as if they were proportional to the dry mass of the test body. Due to the lack of consistency between the data interpretation and the origin of the limit values, it is difficult to evaluate the degree of environmental protection provided by the French acceptance criteria.

9.4 Compliance of tested waste with various criteria

As described in chapter 7, various tests have been carried out on three types monolithic waste, StabW (stabilised APC residue from fluid-bed combustion of RDF), BVR-1 (cement stabilised MSWI fly ash, low fly ash content) and BCR-2 (cement stabilised MSWI fly ash), see also table 7.1. In order to assess the performance of these materials in relation to the criteria proposed in chapter 8.1 and to provide an indirect comparison of the criteria implemented or proposed in the UK, the Netherlands and France, appropriate leaching results for the waste materials are compared to the various criteria below.

Table 9.4 shows the results of batch tests performed on StabW and BCR-1 at L/S = 10 l/kg compared to the corresponding waste acceptance criteria for granular waste (Council Decision 2003/33/EC). The data have been obtained from results of the pH dependence tests performed at the materials' own pH values (Appendix 6). This test is similar to a batch test performed on crushed monolithic materials, except that the material is finer (< 1 mm as opposed to < 4 mm) and the contact time longer (48 hours as opposed to 24 hours) in the pH dependence test. The results of the pH dependence test may therefore be closer to the equilibrium condition sought in both tests, and hence slightly "harsher" than the results of the prescribed batch leaching test (EN 12457-2) would have been. No results were available on fluoride and DOC.

The results in table 9.4 show that both materials comply with the criteria for acceptance of hazardous waste at a landfill for hazardous waste. BCR-1 also complies with the criteria for acceptance of non-hazardous waste and stable, non-reactive hazardous waste at a landfill for non-hazardous waste. For StabW, only the leaching of chloride exceeds the criteria for acceptance of non-hazardous waste and stable, non-reactive hazardous waste at a landfill for non-hazardous waste.

ECN had performed a column leaching test (CEN/TS 14405) on crushed samples of the monolithic waste material BCR-2, and table 9.5 shows the results of the accumulated leached amounts of the measured components at L/S = 2 l/kg and L/S = 10 l/kg together with the acceptance criteria at both L/S values.

Table 9.4 Batch leaching data from pH dependence tests performed on crushed samples of the monolithic waste materials StabW and BRC-1 at L/S = 10 l/kg compared with waste acceptance criteria for granular waste (2003/33/EC).

Component	Unit	L/S = 10 l/kg			
		Batch StabW	Batch BRC-1	WAC	
				non-haz	haz
As	mg/kg	0.18		2	25
Ba	mg/kg	1.8	34	100	300
Cd	mg/kg	0.0007	0.007	1	5
Cr	mg/kg	0.49	0.66	10	70
Cu	mg/kg	0.034	0.033	50	100
Hg	mg/kg	0.019		0.2	2
Mo	mg/kg	0.28	0.22	10	30
Ni	mg/kg	0.03	0.029	10	40
Pb	mg/kg	0.011	2.6	10	50
Sb	mg/kg	0.015	0.069	0.7	5
Se	mg/kg	0.31		0.5	7
Zn	mg/kg	0.005	1.0	50	200
Chloride	mg/kg	19200		15000	25000
Fluoride	mg/kg			60	500
Sulphate	mg/kg	12640	158	20000	50000
DOC	mg/kg			800	1000
Own pH	-	10.7	12.55	-	-

Table 9.5 Results of column leaching test (CEN/TS 14405) performed on a crushed sample of the monolithic waste material BCR-2 (provided by ECN, 2003). Criteria (WAC) for acceptance of granular stable, non-reactive hazardous (and non-hazardous) waste at a landfill for non-hazardous waste and for acceptance of hazardous waste at a landfill for hazardous waste are also shown. Results and criteria are shown both for L/S = 2 l/kg and L/S = 10 l/kg.

Component	Unit	L/S = 2 l/kg			L/S = 10 l/kg		
		Column	WAC		Column	WAC	
			non-haz	haz		non-haz	haz
As	mg/kg						
Ba	mg/kg	0.73	30	100	4	100	300
Cd	mg/kg	0.0027	0.6	3	0.0033	1	5
Cr	mg/kg	0.007	4	25	0.0088	10	70
Cu	mg/kg	0.0099	25	50	0.029	50	100
Hg	mg/kg						
Mo	mg/kg						
Ni	mg/kg	0.0046	5	20	0.021	10	40
Pb	mg/kg	24	5	25	28	10	50
Sb	mg/kg	0.0017	0.2	2	0.0079	0.7	5
Se	mg/kg	-					
Zn	mg/kg	3.5	25	90	4.6	50	200
Chloride	mg/kg	42400	10000	17000	42800	15000	25000
Fluoride	mg/kg						
Sulphate	mg/kg	11400	10000	25000	17800	20000	50000
DOC	mg/kg	57	380	480	69	800	1000

Table 9.5 shows that the results for BCR-2 exceed the limit values for acceptance at a non-hazardous waste landfill for Pb and chloride, both at L/S = 2 l/kg and L/S = 10 l/kg. For sulphate, the limit value for acceptance at a non-hazardous landfill is exceeded for L/S = 2 l/kg, but not for L/S = 10 l/kg. The limit value for acceptance at a landfill for hazardous waste is significantly exceeded for chloride at both L/S values.

Table 9.6 shows the criteria for landfilling of monolithic waste proposed by the UK and set by the Netherlands (from tables 9.1 and 9.3). Both refer to the accumulated leached amounts of components per unit surface area determined over 64 days by the tank leaching test NEN 7345. Such results are available for BCR-2, and the tank leaching results for StabW and BCR-1 presented in Appendix 2 are entirely comparable to the results of NEN 7345 (65.25 days instead of 64 days). These tank leaching results are also shown in table 9.6.

Table 9.6 Criteria for acceptance of monolithic waste from the UK and the Netherlands compared with results of tank leaching tests on the monolithic waste types studied in this project. The criteria and results are 64-days accumulated leached amounts per m² surface area. Results in bold italics exceed the proposed UK criteria for acceptance at a hazardous waste landfill, and results in underlined bold italics exceed the Dutch criteria for acceptance at a hazardous waste landfill

Parameter	UK criteria (proposed)		NL criteria Haz	NMR test results (accumulated release over 64 days in tank leaching test)		
	All units mg/m ² Non-haz	Haz		StabW	BCR-1	BCR-2
As	1.3	20	50	2.9	3.7	-
Ba	45	150	1500	95	33	97
Cd	0.2 (0.03)	1.0 (0.04)	5	0.039	0.034	0.46
Cr	5	25	500	16	17	40
Cu	45	60	500	2.3	0.40	1.4
Hg	0.1 (0.01)	0.4 (0.01)	1	0.42	0.75	-
Mo	7	20	900	1.1	0.54	140
Ni	6	15	400	0.13	0.020	3.5
Pb	6	20	1000	0.72	0.78	31
Sb	0.3	2.5	50	3.6	1.1	19
Se	0.4	5	60	1.8	0.54	-
Zn	30	100	800	1.3	4.5	33
Chloride	10000	20000	250000	223000	36000	<u>880000</u>
Fluoride	60	200	2500	-	-	-
Sulphate	10000	20000	250000	40000	4200	120000
DOC	-	-	-	-	-	-

Table 9.6 shows that the proposed UK criteria for acceptance of monolithic waste at a landfill for hazardous waste are significantly more stringent than those set in the Netherlands. Referring to the same test, the UK criteria for landfilling of hazardous waste are 2.5 to 50 times more stringent than those from the Netherlands. None of the three waste materials complies with the UK criteria for acceptance of monolithic waste at a landfill for hazardous waste, whereas only BCR-2 exceeds the Dutch criteria (and only for one parameter, chloride).

The French criteria for acceptance of monolithic waste at a hazardous waste landfill refer to a one step 24 hour tank leaching test at L/S = 10 l/kg using a test body with a surface area of 126 cm² where the results are reported in mg/kg and compared to the criteria listed in Council Decision 2003/33/EC for granular waste. 24 hour tank leaching data obtained in two steps are available in Appendix 2 for BCR-2 and may be converted to results which are comparable to those that would have been obtained by

the French test. The dry density of the stabilised BCR-2 material is not known, but it is most likely between 1 and 2 kg/l. The results are therefore calculated for dry densities of both 1 kg/l and 2 kg/l and compared to the WAC for granular waste from 2003/33/EC in table 9.7.

Table 9.7 Results of two first stages of tank leaching test NEN 7345 carried out on BCR-2 converted to results that are comparable to the results of the French tank leaching test XP X 31-211 and compared to the WAC for granular hazardous waste in Council Decision 2003/33/EC.

Parameter	Leached in 24 hrs	Leached amount for 126 cm ² and 101 cm ³	Results converted to French test conditions (XP X 31-211)		EU WAC for granular hazardous waste (L/S = 10 l/kg)
			Leached amount based on mass		
			Density = 2 kg/l	Density = 1 kg/l	
	mg/m ²	mg	mg/kg	mg/kg	mg/kg
As					25
Ba	12	0.15	0.75	1.5	300
Cd	0.221	0.0028	0.014	0.028	5
Cr	15.7	0.20	0.98	2.0	70
Cu	0.39	0.0049	0.024	0.049	100
Hg					2
Mo	40.3	0.51	2.5	5.0	30
Ni	0.962	0.012	0.060	0.12	40
Pb	7.11	0.089	0.44	0.89	50
Sb	2.71	0.034	0.17	0.34	5
Se					7
Zn	5.42	0.068	0.34	0.68	200
Chloride	312000	3920	19500	39000	25000
Fluoride					500
Sulphate	64900	816	4100	8100	50000
DOC					1000

Table 9.7 shows that the performance in the French system depends strongly on the dry density of the waste material. It also shows that the results are far below the limit values for all parameters except for chloride which exceeds the limit value for a dry density of 1 l/kg but not for 2 l/kg. BCR-2 will comply with the criteria for landfilling of hazardous waste (for the components measured) if the dry density of the test body is above 1.56 kg/l.

Table 9.8 provides an overview of the degree of compliance of BCR-2 with the criteria for acceptance of hazardous waste at a landfill for hazardous waste in the UK (proposed), the Netherlands, France and with the criteria proposed in this project (under equilibrium-like conditions).

Table 9.8 Overview of the performance of the monolithic waste material BCR-2 in relation to the criteria for acceptance at hazardous waste landfills in various countries and in the system proposed in this project (NMR) under equilibrium-like conditions. + indicates compliance and - indicates non-compliance.

Parameter	United Kingdom (proposed)	Netherlands	France	NMR (this project)
As				
Ba	+	+	+	+
Cd	+ (exceeds most stringent value)	+	+	+
Cr	-	+	+	+
Cu	+	+	+	+
Hg				
Mo	-	+	+	
Ni	+	+	+	+
Pb	-	+	+	+
Sb	-	+	+	+
Se				
Zn	+	+	+	+
Chloride	-	-	exceeds for densities below 1.56 kg/l	-
Fluoride				
Sulphate	-	+	+	+
DOC				

The table shows that BCR-2 consistently fails to fulfil the criteria for chloride in all systems, except possibly the French. It also indicates that the material complies for all other (measured) components in all systems except the UK system, where it also fails for Cr, Mo, Pb, Sb and sulphate.

10. Conclusions and recommendations

“Monolithic waste” is not clearly defined in the Landfill Directive and the Council Decision 2003/33/EC. Most of the existing definitions of “monolithic” waste refer to certain minimum dimensions and physical and mechanical properties of a test body that ensure its integrity over a certain period of time (usually the duration of a so-called tank leaching test).

Generally, “monolithic waste” has been perceived as waste occurring in large blocks or large coherent masses of concrete-like material of substantial strength and very low permeability. It has further been expected that the release of contaminants from monolithic waste material placed in a landfill could be tested and described by means of a tank leaching test performed on test bodies of the monolithic material in question under dynamic (diffusion or release rate controlled) conditions. The results of the study have, however, indicated that waste landfilled as being “monolithic” generally appears to be less “hard” and coherent than originally assumed, and that the composition of the leachate at the bottom of a typical monolithic landfill for most contaminants is likely to be controlled by equilibrium rather than release rate.

The study has further shown that the release behaviour of “monolithic” waste depends not only upon the characteristics of the waste itself (the rate of release and the solubility of the contaminants) but also on the landfill scenario, in particular the geometric surface area per unit volume of the waste material in the landfill, the height of the landfill and rate of infiltration and percolation of precipitation through the landfill. A simplified model has been set up to test whether or not the leachate composition at the base of a landfill containing a given “monolithic” waste material in a given scenario is likely to be controlled by equilibrium or by the rate of release of contaminants from the solid phase. Application of this model on three types of solidified, monolithic waste, for which leaching data were available, indicated that for nearly all leachate components measured, the concentration in the leachate at the base of the landfill would appear to be controlled by equilibrium over a wide range of variation of the above mentioned parameters. This means that the physical structure of the monolithic waste has little effect on the composition of the leachate under the conditions studied and that the leachate composition basically is the same as that which would be expected for the same material in a granular form.

The methodology used by the TAC to develop waste acceptance criteria (WAC) for landfilling of granular waste materials is presented and

discussed in some detail in the report. In order to ensure the same level of environmental protection for landfilling of monolithic waste as for granular waste, the same methodology should be used, only with the source model adjusted (if needed) to fit monolithic waste rather than granular waste. Various model inputs and parameters may be adjusted to fit national conditions, if they differ significantly from those used by the TAC. The same methodology may also be used with local and more specific input data and parameters for site-specific impact assessments.

Based on the above, it is apparent that if the same basic landfill scenarios as those applied by the TAC are used for landfills for hazardous and non-hazardous monolithic waste (or non-hazardous waste landfills receiving stable, non-reactive hazardous monolithic waste), the appropriate level of protection would be achieved if the monolithic waste is crushed to < 4 mm and tested the same way as required for granular waste and the results of the tests are compared to the same WAC as those for granular waste in 2003/33/EC. A waste owner testing his waste would then benefit from the chemical effect of a stabilisation but not from the physical effect, since the latter appears to have little or no effect on the composition of the leachate at the bottom of the landfill under normal conditions. If the water balance is different from that of the TAC modelling, the models could be run again to find adjusted values (for the purpose of illustration, this has actually been done for water balance conditions corresponding to those required in Sweden). Similarly, values could need to be adjusted, possibly by re-running the models, if the downstream requirements are changed (the groundwater quality criteria may be changed or the modelling area may be expanded to include surface water bodies).

It is interesting to notice that during the negotiations in the TAC, the acceptance criteria that actually resulted from the modelling were increased for all parameters (except Cr) to obtain the values listed in 2003/33/EC. The factors by which they were increased range from 1.1 to 17 for non-hazardous waste landfills and from 1.6 to 24 for hazardous waste landfills.

WAC for landfilling of monolithic waste have been set or proposed in the United Kingdom, the Netherlands and France, based on other criteria and test methods (mostly on tank leaching tests). Sufficient test results were available for one monolithic waste material to compare the performance in terms of compliance/non-compliance of the material in the three systems (for hazardous waste landfills) as well as in the system existing for granular waste (based on testing of crushed monolithic material). The comparison shows that the proposed UK criteria are most restrictive whereas the Dutch and the French criteria are more lenient. For the waste tested, the performance was practically the same in the Dutch and French system as well as in the "granular" system, where the material

was crushed prior to testing and evaluated against the criteria for granular waste.

Although the model considerations used are simple and the results should be confirmed by more sophisticated models and by field observations, the following recommendations may be presented:

- In view of the limited information and experience available and the ongoing efforts to develop better test methods and models that better describe the release of components in a monolithic landfill, it is recommended to keep an “open mind”, leaving room for implementation of new developments and improvements.
- At this stage, it is recommended not to distinguish between “granular” and “monolithic” waste materials as such in relation to the setting of waste acceptance criteria landfilling and testing of compliance with these criteria as prescribed in Council Decision 2003/33/EC.
- It is recommended to crush “monolithic” waste (or waste with particle size above 4 to 10 mm) to a particle size < 4 mm and test the leaching properties using the same methods as those prescribed for granular waste (in particular EN 12457-part 1-3 and CEN/TS 14405). CEN/TS 14405, which is a column test, includes the option of testing a material with a particle size < 10 mm provided a large column diameter (10 cm) is used.
- If the TAC scenarios are used in the model calculations without changes, the same WAC as those listed in Council Decision 2003/33/EC for granular waste should apply. If changes are made in input parameters to the models or the scenarios are changes (e.g. in terms of water balances for the landfills or downstream water quality targets and target values), the models must be re-run to obtain new criteria.

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Sammendrag (Summary in Danish)

I EU's Rådsbeslutning af 19. december 2002 om opstilling af kriterier og procedurer for modtagelse af affald på deponeringsanlæg i henhold til Artikel 16 og Bilag II i Direktiv 1999/31/EF om deponering af affald (2003/33/EF) er der sat kriterier og grænseværdier for modtagelse af granulært affald på visse kategorier og underkategorier af deponeringsanlæg. Der er ikke sat specifikke kriterier for modtagelse af monolitisk affald på EU-niveau, men det er anført, at medlemsstaterne på nationalt niveau skal sætte kriterier for modtagelse af monolitisk affald til deponering. Det er yderligere anført, at disse kriterier for en given kategori eller underkategori af deponeringsanlæg skal give det samme miljøbeskyttelsesniveau, som de gældende kriterier for granulært affald. I Rådsbeslutningen 2003/33/EF stilles der krav om opstilling af acceptkriterier for monolitisk affald for deponeringsanlæg for farligt affald og for anlæg til deponering af ikke-farligt affald, som modtager stabilt, ikke-reaktivt farligt affald.

En opfyldelse af dette forudsætter, at modtagelseskriterierne for monolitisk affald fastlægges på grundlag af de samme scenariebaserede modelberegninger, som blev anvendt til fastlæggelse af modtagekriterierne for granulært affald. Det vil dog være nødvendigt at tilpasse deponerings-scenariet til et deponi for monolitisk affald, og kildestyrkebeskrivelsen skal tilpasses udvaskningen af forureningskomponenter fra monolitisk affald, hvilket kan afvige fra beskrivelsen udvaskning fra granulært affald.

Med henblik på at understøtte implementeringen af Rådsbeslutningens krav om opstilling af kriterier for modtagelse af monolitisk affald på deponeringsanlæg i de nordiske lande har DHI – Institut for Vand og Miljø (DHI) i Danmark i samarbejde med Statens geotekniske institut (SGI) i Sverige og Statens Tekniske Forskningscentral (VTT) i Finland gennemført et projekt finansieret af Nordisk Ministerråd. Projektet har haft følgende hovedformål:

- At udvikle en metode til fastlæggelse af kriterier for modtagelse af monolitisk affald på deponeringsanlæg i overensstemmelse med kravene i EU's rådsbeslutning 2003/33/EF.
- At give anbefalinger vedrørende passende metoder til testning af monolitisk affald og forslag til tilhørende kriterier/grænseværdier for modtagelse af monolitisk affald på deponeringsanlæg. Disse kriterier skal medføre samme niveau af miljøbeskyttelse, som de

i Rådsbeslutning 2003/33/EF anførte kriterier for modtagelse af granulært affald.

De væsentligste resultater og konklusioner af projektet, som blev gennemført i perioden 2003 – 2005, er opsummeret i det følgende:

”Monolitisk affald” er hverken i EU’s Deponeringsdirektiv 1999/31/EF eller i Rådsbeslutningen 2003/33/EF klart defineret. De fleste eksisterende definitioner af ”monolitisk” affald eller materiale refererer til visse minimumskrav til fysiske dimensioner og mekanisk styrke for et testlegeme for at sikre, at dette bevarer sin integritet i en periode (oftest i forbindelse med gennemførelsen af en såkaldt tankudvaskningstest).

”Monolitisk affald” er generelt blevet opfattet som affald, der forekommer som store blokke eller store, sammenhængende masser af betonlignende materiale af betydelig mekanisk styrke og meget lav hydraulisk ledningsevne. Det har endvidere implicit været forventet, at stofudvaskningen fra monolitisk materiale placeret i et deponi skulle kunne testes og beskrives ved hjælp af en tankudvaskningstest udført på prøvelegemer af materialet under dynamiske betingelser (diffusionskontrolleret udvaskning). Resultaterne af dette projekt har dog vist, at affald, der er deponeret som værende ”monolitisk”, i mange tilfælde viser sig at være mindre ”hårdt” og mindre sammenhængende end oprindeligt antaget, og at sammensætningen af perkolatet ved bunden af et typisk deponeringsanlæg for monolitisk affald for de fleste forureningskomponenters vedkommende ofte synes at være ligevægtskontrolleret snarere end styret af frigivelseshastigheden fra den faste fase (diffusionskontrolleret).

Projektet har endvidere vist, at stofudvaskningen fra ”monolitisk” affald ikke alene afhænger af karakteren af affaldet selv (frigivelseshastigheden og opløseligheden af forureningskomponenterne), men også af deponeringsscenariet, herunder specielt den geometriske overflade per volumenenhed affald i deponiet, højden af deponiet og den hastighed, hvormed nedbør infiltrerer i og perkolerer gennem affaldet. Der er udviklet en simplificeret udvaskningsmodel for et monolitisk affaldsdeponi til testning af, om perkolatet ved bunden af en given type ”monolitisk” affald i et givet deponeringsscenarie primært synes ligevægtskontrolleret eller primært synes styret af hastigheden af stoffrigivelsen fra den faste fase. Anvendelse af modellen på tre forskellige typer solidificeret, monolitisk affald, for hvilke der forelå anvendelige udvaskningsdata, indikerede, at sammensætningen af perkolatet ved bunden af deponiet for en betydelig variationsbredde af de ovennævnte parametre syntes at være ligevægtskontrolleret. Dette betyder, at den fysiske struktur af det monolitiske affald har en meget begrænset indflydelse på sammensætningen af perkolatet under de anvendte betingelser, og at perkolatets sammensætning som funktion af tiden stort set er den samme som kunne forventes for det samme materiale, hvis det var på granulær form.

Den metodik, som blev anvendt af den Tekniske Adaptationskomité (TAC) ved udviklingen af de i Rådsbeslutning 2003/33/EF anførte acceptkriterier for deponering af granulært affald præsenteres og diskuteres i denne rapport. For at sikre samme miljømæssige beskyttelsesniveau for deponering af monolitisk affald som ved deponering af granulært affald bør der i begge tilfælde anvendes samme metodologi, dog således at kildestyrkedelen af modellen for monolitisk affald (om nødvendigt) er tilpasset de specielle forhold, som måtte være gældende herfor. Desuden kan det være nødvendigt at tilpasse forskellige input og parametre til nationale forhold, såfremt de afviger væsentligt fra dem, der blev anvendt af TAC. Den samme metodik kan også anvendes med mere lokale og mere specifikke inputdata og parametre ved stedspecifikke vurderinger af miljøpåvirkningen.

På grundlag af ovenstående kan det konkluderes, at hvis man for deponier for farligt og for ikke-farligt monolitisk affald (eller for deponeringsanlæg for ikke-farligt affald, som modtager stabilt, ikke-reaktivt monolitisk affald) anvender de samme grundlæggende deponeringsscenerier, som blev anvendt af TAC, så vil man opnå samme grad af miljøbeskyttelse i begge tilfælde, hvis det monolitiske affald nedkneses til < 4 mm og testes på samme måde, som foreskrevet for det granulære affald i 2003/33/EF, og hvis man anvender samme grænseværdier, som anført for granulært affald i 2003/33/EF. En affaldsproducent, som efter stabilisering/solidificering tester sit monolitiske affald, ville således kunne drage fordel af en eventuel kemisk stabilisering af affaldet, men ikke af den eventuelle fysiske effekt hidrørende fra strukturen, da sidstnævnte under normale omstændigheder synes at have ringe eller ingen effekt på sammensætningen af perkolatet ved bunden af et deponi. Hvis vandbalancen afviger fra den, som blev anvendt ved TAC-modelleringen, bør modellerne køres igen med den ændrede vandbalance (dette er faktisk blevet gjort i projektet for vandbalancer svarende til kravene i den nationale implementering af 2003/33/EF i Sverige). Tilsvarende justeringer kunne nødvendiggøres af eventuelle ændringer i de nedstrøms modelforudsætninger (f.eks. ved ændring i grundvandskvalitetskravene eller ved udvidelse af modelområdet til også at omfatte overfladevand).

Det er interessant at bemærke, at de acceptkriterier for deponering af granulært affald, som er anført i 2003/33/EF, og som er fastlagt med udgangspunkt i modelberegninger, ved de efterfølgende politiske diskussioner i TAC'en (med Cr som eneste undtagelse) er blevet forhøjet i forhold til resultaterne af modelberegningerne med faktorer, der varierer fra 1,1 til 17 for deponeringsanlæg for ikke-farligt affald, som modtager stabilt, ikke-reaktivt farligt affald, og fra 1,6 to 24 for deponeringsanlæg for farligt affald.

Der er fastsat eller foreslået kriterier for modtagelse af monolitisk affald til deponering i England, Holland og Frankrig, baseret på andre principper og testmetoder (primært tankudvaskningstests) end de her foreslå-

ede. For ét monolitisk affaldsmateriale forelå der tilstrækkeligt mange udvaskningsdata til, at det var muligt at sammenligne, hvorledes det ville klare sig i de tre forskellige landes systemer (for deponeringsanlæg til farligt affald) samt i forhold til kriterierne for deponering af farligt, granulært affald (baseret på testning af nedknust monolitisk materiale). Sammenligningen viste, at de foreslåede engelske kriterier, der er baseret på modelberegninger, som ikke tager hensyn til opløselighedsbegrænsninger af stofudvaskningen, er de mest restriktive, mens de hollandske og franske kriterier er mindre stringente. De franske kriterier er videnskabeligt dårligt funderet, hvilket blandt andet betyder, at overholdelse af visse grænseværdier afhænger af materialets densitet. For den testede affaldstype var resultatet i det hollandske og franske system og i det ”granulære” system baseret på nedknust prøve stort set det samme.

Selv om de anvendte modelberegninger er simplificerede, og resultaterne bør understøttes og bekræftes af mere sofistikerede modeller og et mere omfattende datagrundlag samt af feltobservationer, kan der gives følgende anbefalinger:

- I lyset af den begrænsede information og de begrænsede erfaringer på området samt de igangværende bestræbelser på at forbedre testmetoder og modeller til beskrivelse af udvaskningen af forureningskomponenter i et monolitisk deponeringsanlæg anbefales det, at man indtil videre bevarer et åbent sind og giver plads til implementering af nye udviklinger og forbedringer.
- På det nuværende stade anbefales det, at man ikke skelner mellem ”granulært” og ”monolitisk” affald som sådan i forhold til fastsættelse af acceptkriterier for deponering og testning af overensstemmelse med disse kriterier som foreskrevet i Rådsbeslutning 2003/33/EF.
- Det anbefales at nedknuse ”monolitisk” affald (eller affald med partikelstørrelse over 4 – 10 mm) til en partikelstørrelse < 4 mm og at teste udvaskningsegenskaberne ved hjælp af de samme metoder, som er foreskrevet for granulært affald (specielt 12457, part 1-3 og EN/TS 14405). EN/TS 14405, som er en kolonneudvaskningstest, omfatter en mulighed for testning af et materiale med partikelstørrelse < 10 mm, såfremt man anvender en kolonne med diameter på 10 cm.
- Hvis TAC-scenarierne anvendes i modelberegningerne uden ændringer, bør de samme acceptkriterier som de i Rådsbeslutning 2003/33/EF anførte for granulært affald, også anvendes for monolitisk affald testet på ovennævnte måde. Hvis der er afvigelser i forhold hertil i input-parametrene eller scenarierne (f.eks. i vand-

balancerne eller de nedstrøms kvalitetskriterier til grundvand eller overfladevand), vil det være nødvendigt at køre modellerne igen under de ændrede betingelser for at beregne nye kriterier.

Appendix 1

Standardisation work on leaching tests for monolithic waste

Background

Standardised test methods for leaching studies and chemical analysis for waste materials are developed in the technical committee TC 292 “Characterization of waste” under the European Committee for Standardisation (CEN). The standardisation work on leaching tests for monolithic waste is divided between two working groups (WGs): one (WG2 – Leaching test procedures) focusing on compliance testing and the other (WG6 - Basic characterisation tests for leaching behaviour) on characterisation test for understanding of the leaching behaviour of waste. Standardisation work on the compliance leaching test for monolithic material started already in 1996 and on the basic characterisation test in year 2001.

The standard EN 12920 on the methodology for determination of leaching behaviour of waste under specified conditions has been used in the development of a characterisation leaching test for monolithic waste. Scenarios of interest and related environmental conditions, materials of concern, main leaching factors and operational aspect of existing procedures are considered in the test development. The main waste stream of interest among several member states is cement stabilised or vitrified air pollution control (APC) residues from municipal solid waste incineration, i.e. fly ash with or without the residues from the acid gas cleaning process. Other materials expected to be important in the future are waste-based building products falling under Construction Products Directive, for which information is needed about the emissions of regulated dangerous substances to soil, groundwater and surface water.

The development of a leaching method for monolithic waste has appeared to be more complex than for granular waste. A recent study at ECN on the modelling of the leaching behaviour has also brought up new information on relevant aspects (for example solubility limits of certain compounds, e.g. sulphate) to be considered in the test development. Several competing reactions (e.g. precipitation), matrix properties (e.g. depletion, porosity) and also environmental conditions (e.g. scenarios) highly influence the leaching behaviour of monolithic waste.

The late start on the basic characterisation test also hindered the standardisation work on the compliance test for monolithic waste, because the compliance test needs to be linked to basic characterisation test. In fact there are different views on the purpose of the compliance test among the CEN member states. Some member states want a compliance method that is also suitable for process control, which means that the test method must be very quick and simple. This may be in conflict with the linkage to the characterisation test.

The monolithic character of the waste also needs to be assessed for testing of the waste as a monolithic rather than a granular material.

The progress in the development of the standardisation work in CEN/TC 292 on basic characterisation leaching methods for monolithic waste is briefly reviewed in the following.

Development of basic characterization test

The scope of the work item for WG6 is to specify a test method for determining the leaching behaviour of monolithic waste under dynamic conditions. The aim of the test is to provide information on release of inorganic constituents from a regularly shaped specimen of monolithic wastes. The test can also be used to determine the dominant release mechanisms. The test method under development is named “Dynamic leaching test for monolithic waste” or “Dynamic monolithic leaching test (DMLT)”.

Four typical situations to be addressed according to the DMLT are identified in the test development work. They are described as follows:

Situation 1: The so-called “pillar in the canal”: Material continuously exposed to high water flow and therefore largely influenced by the surroundings and not predominantly affected by the constituents of the material already released.

Situation 2: The so-called “pillar in the groundwater”: Material continuously exposed to a limited water flow in which the material itself imposes the leaching conditions, and in which the constituents of the material already released may result in a modification of the release of other constituents.

Situation 3: The so-called “stabilised waste monofill”: Material continuously exposed to an extreme low water flow in which leaching conditions are fully controlled by constituents of the material already released resulting in a release largely controlled by equilibrium.

Situation 4: The so-called “stabilized embankment or road base concrete”: Material in open air conditions and exposed to intermittent water contact affecting the release of the constituents of the material.

For the standardisation of test methods for landfill scenarios, situations 2 and 3 are most relevant. Especially the equilibrium phase in situation 3 is considered important and needs to be part of the test procedure. In situation 3, diffusion controlled leaching will probably not occur. The starting point in the standardisation work is to develop a common testing scheme that covers all the situations. The testing scheme may then be modified according to the actual scenario and waste material. A common testing scheme for a DMLT test is to be performed according to the following

principle: A regularly shaped test specimen is immersed in water (leachant amount (L) depending on surface (A)) and the water is renewed with certain time intervals (and analysed).

Three different procedural phases in the testing scheme are pointed out:

- The initial ‘wash-off’ phase: Intended to eliminate short term artificial effects that should not be included for the prediction (e.g. removal of dust, precipitation). This step is not always necessary for all materials.
- The ‘equilibration’ phase: Intended to approach equilibrium between solid and liquid and stabilise the system.
- The ‘time dependent leaching’ phase: Intended to determine the time dependent leaching behaviour of the material and identify the leaching controlling mechanisms (diffusion controlled release, solubility controlled release and other situations).

The appropriate testing conditions for different scenarios and test materials are different. For vitrified waste materials probably only phase 1 is relevant. Phase 2 is most relevant for porous materials, for which the objective is to approach equilibrium within a reasonable period of time. For non-porous materials this may not be relevant. Furthermore it is required that the leachant used in phase 3 shall be the same as the leachant in phase 2.

Testing conditions to be considered in the different situations are as follows:

- Situation 1: No different situations for the time being (not environmentally relevant).
- Situation 2: Two different situations may be identified:
 - 2a) No external control on the leaching behaviour of the material. The material imposes the conditions of the leaching. For reactive materials the leaching test can be performed either on fresh materials or on altered material (carbonated, oxidised).
 - 2b) The material is exposed to the leaching conditions imposed by the environment (e.g. CO₂ control or control by composition of leachant).
- Situation 3: Different procedure, with low L/A ratio also in phase 3, and probably lower renewal rate and/or longer duration.
- Situation 4: Discussion on required changes in test procedure has been postponed because of the complex character of this situation.

An experimental study on the influence of selected parameters on the leaching behaviour in the DMLT was carried out by members of CEN/TC

292 WG6, including DHI and VTT, in 2004 in order to collect information on testing conditions and test performance. Some of the parameters studied are:

- The effect of variation of L/A and contact time in phase 1
- The effect of variation of L/A in phase 2
- The effect of variation of contact time in phase 2
- The effect of using either agitation or circulation of leachant using a pump
- The effect of variation of the composition of the leachant (default: demineralised water)

A few of the results of the study (including results produced by DHI and VTT) are shown in Appendix 2, and some of these have been used in Chapter 7 and Chapter 9.

In the study, special attention was paid to the time needed for achieving equilibrium in phase 2. The change in conductivity was followed closely during phase 2 in order to check whether this parameter could be used as an indicator for the equilibrium status. The results from the experimental study clearly showed that equilibrium was not achieved under the selected testing conditions. Further modelling based on the results indicated that the L/A-ratio in the test procedure needs to be significantly lower (L/A about 0.1) for equilibrium conditions to appear within a practical time period. This means that the DMLT cannot be used for assessment of the release at equilibrium.

Further work

Based on the experience from the experimental work there are now two possible approaches for further work:

For assessment of the release in scenarios with equilibrium conditions: There are already existing tools such as the batch test for granular waste EN 12457 and the percolation test for granular material TS 14405.

For assessment of the release in scenarios where diffusion is the dominant leaching controlling mechanism: A DMLT will be developed based on the results from the experimental work. The field of application of the DMLT needs to be clearly defined.

Appendix 2

Results of leaching tests carried out on stabilised wastes

Experimental test of tank leaching procedure (CEN/TC 292/WG6)

DHI/2004

Material: StabW, stabilised waste from RagnSells

DHI-ID: R-235-04A

Side a:	7.7	cm	Area:	365.02	cm ²
Side b:	7.8	cm	Volume:	474.47	cm ³
Side c:	7.9	cm	Weight:	635.57	g

Fraction	Contact time days	Accum. time days	Volume ml	L/A ml/cm ²	pH	Conductivity mS/m
E1	0.25	0.25	726	2.0	10.67	260
E2	3.0	3.25	730	2.0	10.74	140
E3-1	2	5.25	2920	8.0	10.51	180
E3-2	5	10.25	2920	8.0	10.54	160
E3-3	7	17.25	2919	8.0	10.29	120
E3-4	1	18.25	2921	8.0	10.28	29
E3-5	19	37.25	2919	8.0	10.20	120
E3-6	28	65.25	2920	8.0	10.15	96

Measured concentrations in eluates:

Fraction	As mg/l	Ba mg/l	Cd mg/l	Cr mg/l	Cu mg/l	Hg mg/l	Mo mg/l	Ni mg/l
E1	0.00825	0.23	0.000024	0.0294	0.0014	0.001257	0.0042	0.000391
E2	0.018317	0.45	0.000049	0.075963	0.027	0.001033	0.016	0.002192
E3-1	0.005515	0.16	0.000035	0.02557	0.0027	0.001076	0.0025	0.000321
E3-2	0.005123	0.18	0.000021	0.023834	0.0015	0.001134	0.0021	0.000301
E3-3	0.005134	0.18	0.000019	0.021699	0.0012	0.00111	0.0015	0.000215
E3-4	0.003513	0.08	0.000017	0.01626	0.00067	0.001165	0.00058	0.000014
E3-5	0.005681	0.21	0.000127	0.07031	0.015	0.000098	0.00066	0.000039
E3-6	0.004194	0.21	0.000247	0.015193	0.00062	0.000149	0.0011	0.000087

Fraction	Pb mg/l	Sb mg/l	Se mg/l	Zn mg/l	Chloride mg/l	Fluoride mg/l	Sulphate mg/l	DOC mg/l
E1	0.0015	0.015464	0.005341	0.0005	690		120	
E2	0.0016	0.006357	0.022042	0.0005	4200		380	
E3-1	0.0014	0.0076	0.004155	0.002	450		69	
E3-2	0.0018	0.00774	0.001697	0.0005	410		77	
E3-3	0.0016	0.007666	0.004808	0.001	300		66	
E3-4	0.0009	0.006711	0.000844	0.001	43		17	
E3-5	0.00053	0.00265	0.000038	0.007	280		81	
E3-6	0.0020	0.007313	0.004333	0.004	87		61	

Experimental test of tank leaching procedure (CEN/TC 292/WG6)

DHI/2004

Material: StabW, stabilised waste from RagnSells

DHI-ID: R-235-04A

Side a:	7.7	cm	Area:	365.02	cm ²
Side b:	7.8	cm	Volume:	474.474	cm ³
Side c:	7.9	cm	Weight:	635.57	g

Fraction	Contact time days	Accum. time days	Volume ml	L/A ml/cm ²	pH	Conductivity mS/m
E1	0.25	0.25	726	2.0	10.67	260
E2	3.0	3.25	730	2.0	10.74	140
E3-1	2	5.25	2920	8.0	10.51	180
E3-2	5	10.25	2920	8.0	10.54	160
E3-3	7	17.25	2919	8.0	10.29	120
E3-4	1	18.25	2921	8.0	10.28	29
E3-5	19	37.25	2919	8.0	10.20	120
E3-6	28	65.25	2920	8.0	10.15	96

Calculated release during each step:

Fraction	As mg/m ²	Ba mg/m ²	Cd mg/m ²	Cr mg/m ²	Cu mg/m ²	Hg mg/m ²	Mo mg/m ²	Ni mg/m ²
E1	0.164	4.57	0.00048	0.585	0.0278	0.0250	0.0835	0.00778
E2	0.366	9.00	0.00098	1.519	0.5400	0.0207	0.3200	0.04384
E3-1	0.441	12.80	0.00280	2.045	0.2160	0.0861	0.2000	0.02568
E3-2	0.410	14.40	0.00168	1.907	0.1200	0.0907	0.1680	0.02408
E3-3	0.411	14.39	0.00152	1.735	0.0960	0.0888	0.1200	0.01719
E3-4	0.281	6.40	0.00136	1.301	0.0536	0.0932	0.0464	0.00112
E3-5	0.454	16.79	0.01016	5.623	1.1995	0.0078	0.0528	0.00312
E3-6	0.336	16.80	0.01976	1.215	0.0496	0.0119	0.0880	0.00696

Fraction	Pb mg/m ²	Sb mg/m ²	Se mg/m ²	Zn mg/m ²	Chloride mg/m ²	Fluoride mg/m ²	Sulphate mg/m ²	DOC mg/m ²
E1	0.0298	0.308	0.106	0.00994	13724	0	2387	0
E2	0.0320	0.127	0.441	0.01000	83995	0	7600	0
E3-1	0.1120	0.608	0.332	0.15999	35998	0	5520	0
E3-2	0.1440	0.619	0.136	0.04000	32798	0	6160	0
E3-3	0.1279	0.613	0.384	0.07997	23990	0	5278	0
E3-4	0.0720	0.537	0.068	0.08002	3441	0	1360	0
E3-5	0.0424	0.212	0.003	0.55978	22391	0	6477	0
E3-6	0.1600	0.585	0.347	0.31998	6960	0	4880	0

Experimental test of tank leaching procedure (CEN/TC 292/WG6)

DHI/2004

Material: StabW, stabilised waste from RagnSells

DHI-ID: R-235-04A

Side a:	7.7	cm	Area:	365.02	cm ²
Side b:	7.8	cm	Volume:	474.474	cm ³
Side c:	7.9	cm	Weight:	635.57	g

Fraction	Contact time days	Accum. time days	Volume ml	L/A ml/cm ²	pH	Conductivity mS/m
E1	0.25	0.25	726	2.0	10.67	260
E2	3	3.25	730	2.0	10.74	140
E3-1	2	5.25	2920	8.0	10.51	180
E3-2	5	10.25	2920	8.0	10.54	160
E3-3	7	17.25	2919	8.0	10.29	120
E3-4	1	18.25	2921	8.0	10.28	29
E3-5	19	37.25	2919	8.0	10.2	120
E3-6	28	65.25	2920	8.0	10.15	96

Accumulated release:

Fraction	As mg/m ²	Ba mg/m ²	Cd mg/m ²	Cr mg/m ²	Cu mg/m ²	Hg mg/m ²	Mo mg/m ²	Ni mg/m ²
E1	0.1640869	4.574544	0.000477	0.584746	0.027845	0.025000877	0.083535	0.007777
E2	0.5304068	13.57405	0.001457	2.103923	0.567815	0.045659745	0.403518	0.051614
E3-1	0.9715827	26.37335	0.004257	4.149411	0.783804	0.131735028	0.603507	0.077293
E3-2	1.3814002	40.77256	0.005937	6.056026	0.903797	0.222450058	0.771497	0.101372
E3-3	1.791957	55.16684	0.007456	7.791257	0.999759	0.311214783	0.89145	0.118565
E3-4	2.0730779	61.56868	0.008817	9.092431	1.053374	0.404441592	0.937863	0.119685
E3-5	2.5273773	78.36201	0.018973	14.715	2.252898	0.412278478	0.990642	0.122804
E3-6	2.862879	95.16109	0.038732	15.93037	2.302495	0.424197825	1.078637	0.129763

Fraction	Pb mg/m ²	Sb mg/m ²	Se mg/m ²	Zn mg/m ²	Chloride mg/m ²	Fluoride mg/m ²	Sulphate mg/m ²	DOC mg/m ²
E1	0.029834	0.307568	0.106229	0.009945	13723.63	0	2386.719	0
E2	0.0618322	0.434701	0.547045	0.019944	97719.03	0	9986.302	0
E3-1	0.1738261	1.042668	0.879426	0.179935	133717.1	0	15506	0
E3-2	0.3178182	1.661834	1.015179	0.219933	166515.3	0	21665.66	0
E3-3	0.4457674	2.274871	1.399666	0.299901	190505.7	0	26943.56	0
E3-4	0.5177881	2.811905	1.467206	0.379924	193946.7	0	28303.96	0
E3-5	0.5601712	3.023821	1.470244	0.939702	216337.8	0	34781.38	0
E3-6	0.7201625	3.608829	1.816865	1.259684	223297.4	0	39661.11	0

Experimental test of tank leaching procedure (CEN/TC 292/WG6)

DHI/2004

Material: StabW, stabilised waste from RagnSells

DHI-ID: R-235-04A

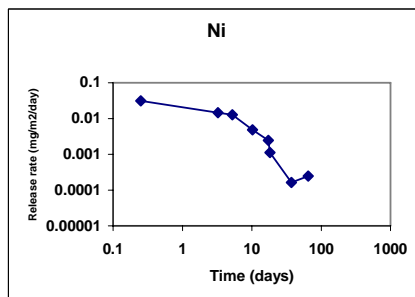
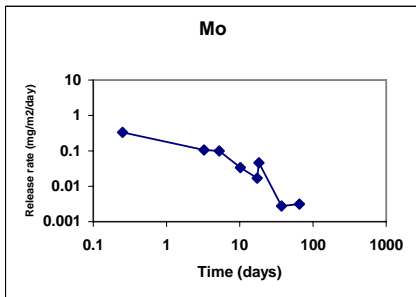
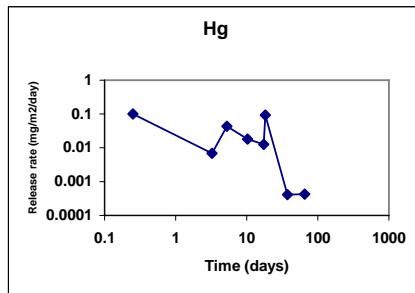
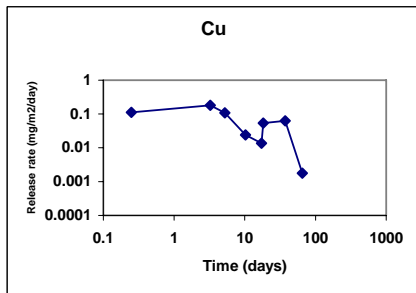
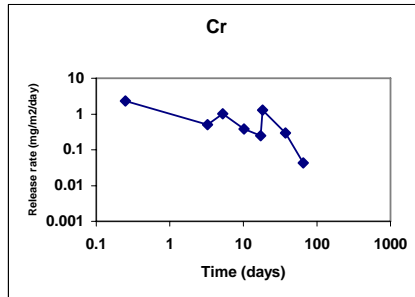
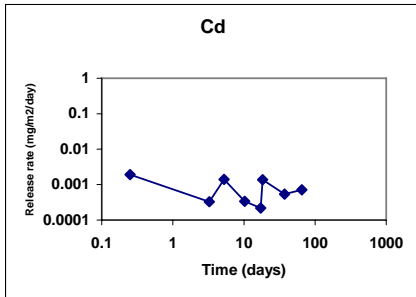
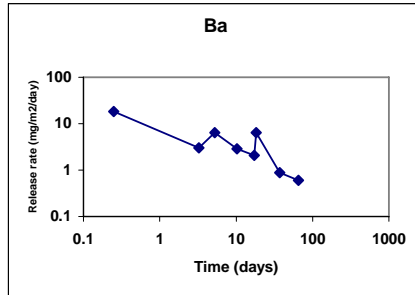
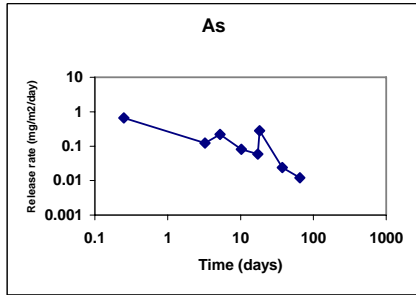
Side a:	7.7	cm	Area:	365.02	cm ²
Side b:	7.8	cm	Volume:	474.474	cm ³
Side c:	7.9	cm	Weight:	635.57	g

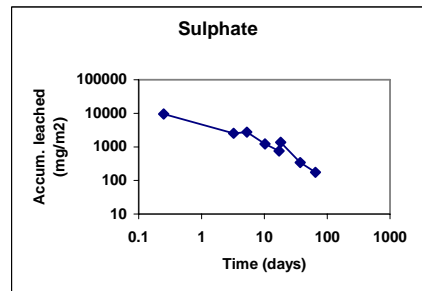
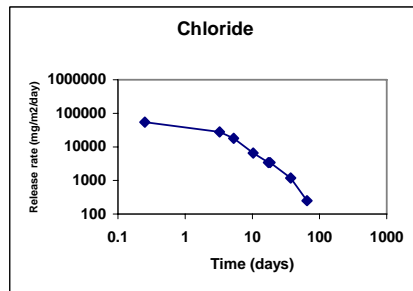
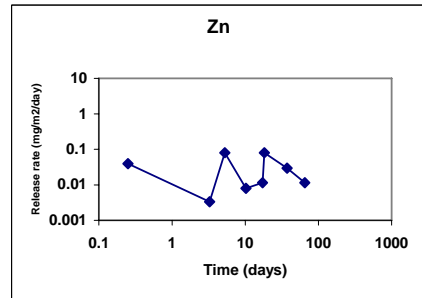
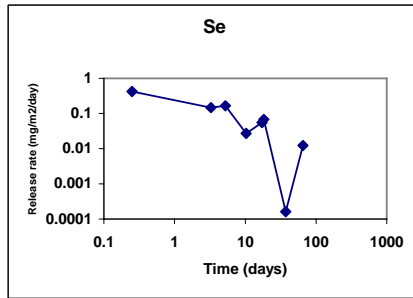
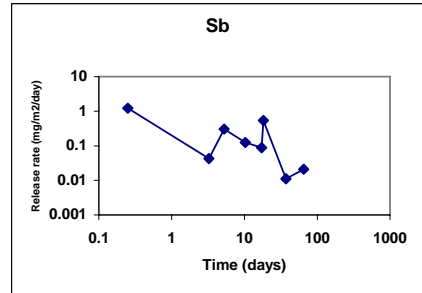
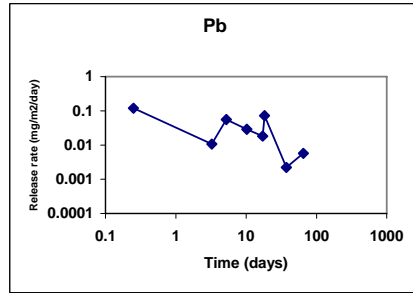
Fraction	Contact time days	Accum. time days	Volume ml	L/A ml/cm ²	pH	Conductivity mS/m
E1	0.25	0.25	726	2.0	10.67	260
E2	3	3.25	730	2.0	10.74	140
E3-1	2	5.25	2920	8.0	10.51	180
E3-2	5	10.25	2920	8.0	10.54	160
E3-3	7	17.25	2919	8.0	10.29	120
E3-4	1	18.25	2921	8.0	10.28	29
E3-5	19	37.25	2919	8.0	10.2	120
E3-6	28	65.25	2920	8.0	10.15	96

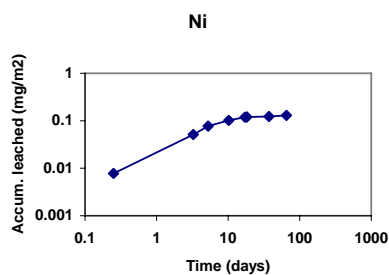
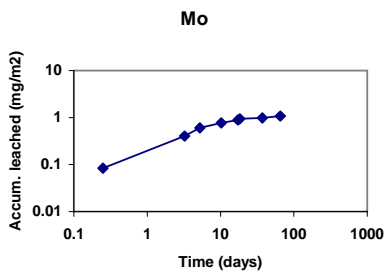
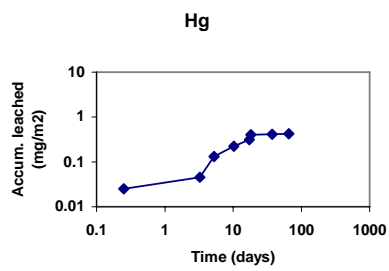
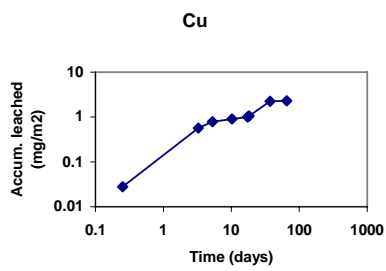
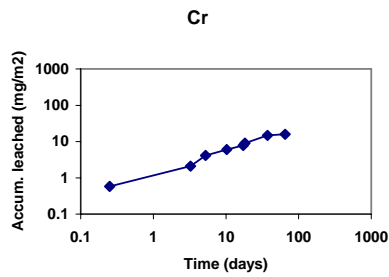
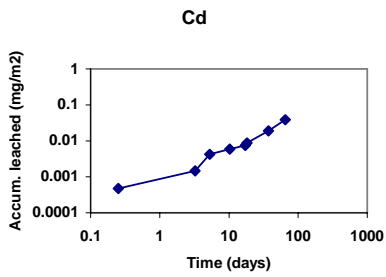
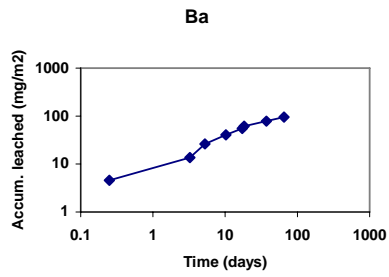
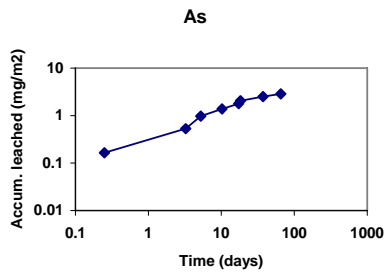
Release rates for each fraction:

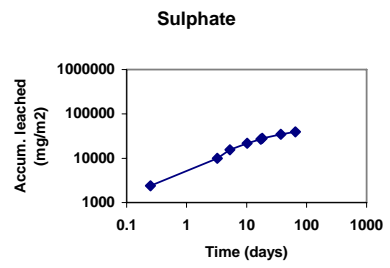
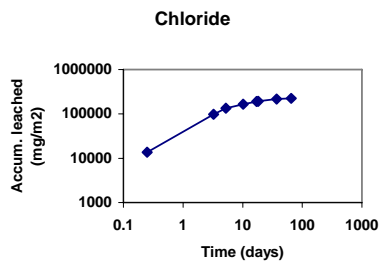
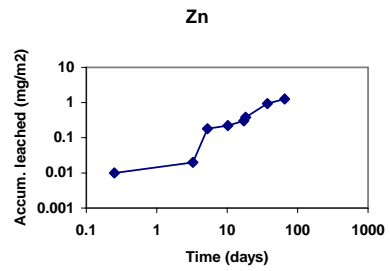
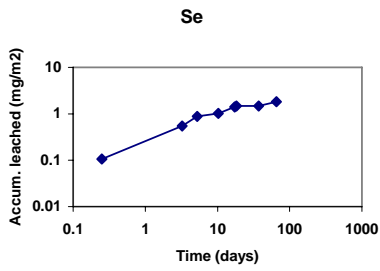
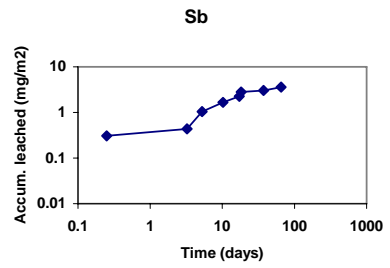
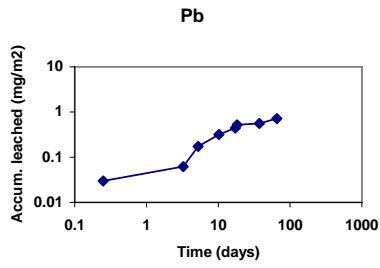
Fraction	As mg/m ² /d	Ba mg/m ² /d	Cd mg/m ² /d	Cr mg/m ² /d	Cu mg/m ² /d	Hg mg/m ² /d	Mo mg/m ² /d	Ni mg/m ² /d
E1	0.6563476	18.29818	0.001909	2.338984	0.11138	0.100003507	0.334141	0.031107
E2	0.1221066	2.999836	0.000327	0.506392	0.17999	0.006886289	0.106661	0.014613
E3-1	0.2205879	6.399649	0.0014	1.022744	0.107994	0.043037642	0.099995	0.012839
E3-2	0.0819635	2.879842	0.000336	0.381323	0.023999	0.018143006	0.033598	0.004816
E3-3	0.058651	2.056326	0.000217	0.24789	0.013709	0.012680675	0.017136	0.002456
E3-4	0.2811208	6.401841	0.00136	1.301174	0.053615	0.093226809	0.046413	0.00112
E3-5	0.0239105	0.883859	0.000535	0.295925	0.063133	0.000412468	0.002778	0.000164
E3-6	0.0119822	0.599967	0.000706	0.043406	0.001771	0.000425691	0.003143	0.000249

Fraction	Pb mg/m ² /d	Sb mg/m ² /d	Se mg/m ² /d	Zn mg/m ² /d	Chloride mg/m ² /d	Fluoride mg/m ² /d	Sulphate mg/m ² /d	DOC mg/m ² /d
E1	0.1193359	1.230274	0.424915	0.039779	54895	0	9547	0
E2	0.0106661	0.042378	0.146939	0.003333	27998	0	2533	0
E3-1	0.0559969	0.303983	0.166191	0.079996	17999	0	2760	0
E3-2	0.0287984	0.123833	0.027151	0.008	6560	0	1232	0
E3-3	0.0182785	0.087577	0.054927	0.011424	3427	0	754	0
E3-4	0.0720207	0.537034	0.067539	0.080023	3441	0	1360	0
E3-5	0.0022307	0.011153	0.00016	0.029462	1178	0	341	0
E3-6	0.005714	0.020893	0.012379	0.011428	249	0	174	0









Results of pH-static leaching test

DHI/2004

Material: StabW, stabilised waste from RagnSells

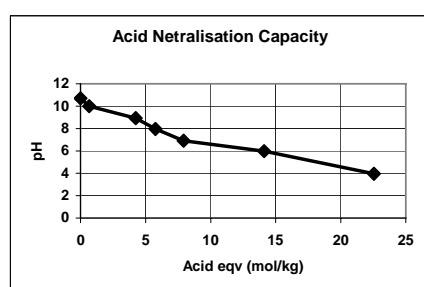
DHI-ID: R-235-04A

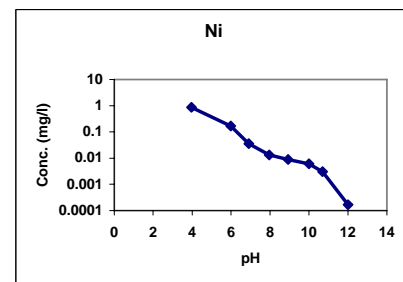
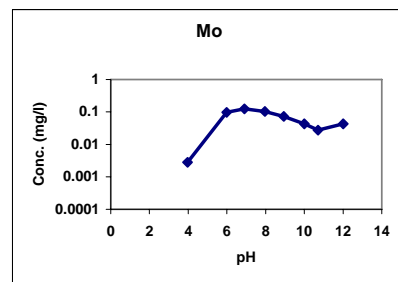
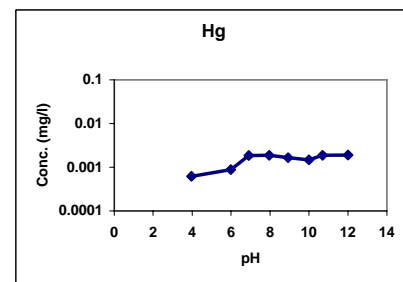
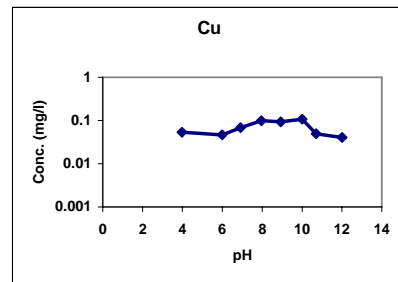
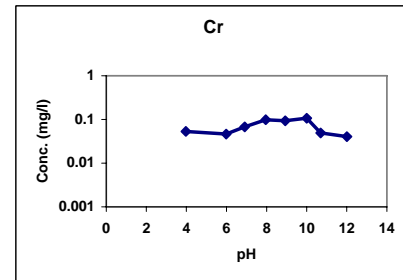
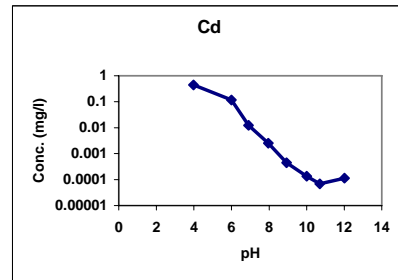
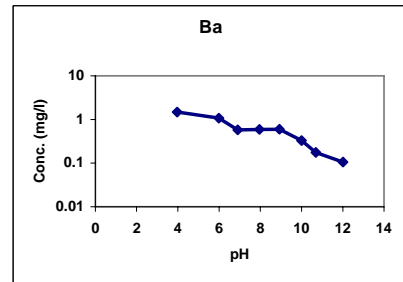
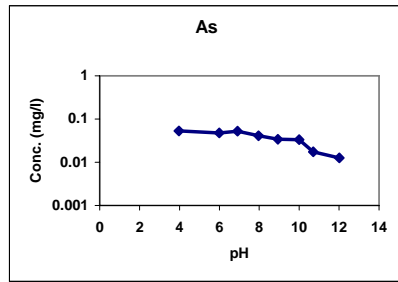
Sample ID	pH	Test material g DM	Liquid total ml	L/S l/kg	Volume acid ml	Volume base ml	Normality mol/l	ANC mol/kg
R235 pH4	3.97	40.01	417.83	10.4	180.38		5	22.54
R235 pH6	5.99	40.01	398.90	10.0	112.88		5	14.11
R235 pH7	6.91	40.04	428.77	10.7	63.41		5	7.92
R235 pH8	7.96	40.04	424.90	10.6	46.02		5	5.75
R235 pH9	8.93	40.01	404.43	10.1	33.91		5	4.24
R235 pH10	10.00	40.01	400.00	10.0	13.28		2	0.66
R235 Own	10.70	40.01	400.00	10.0	0	0		0.00
R235 pH12	12.01	40.04	400.06	10.0		28.88	1	-0.72

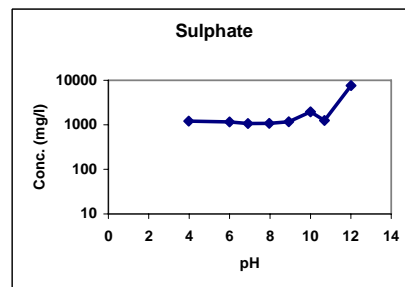
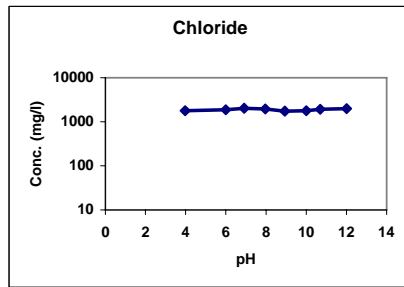
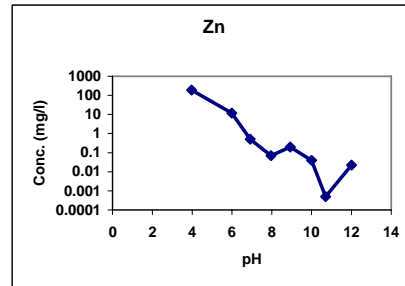
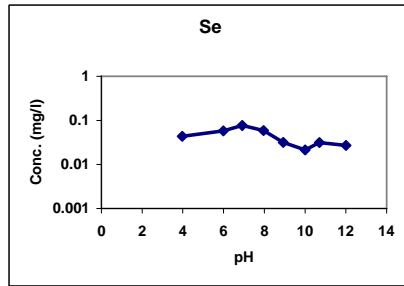
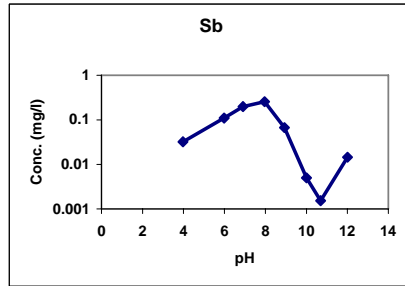
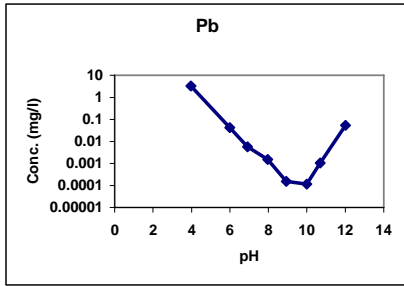
Concentration in eluates:

pH	As mg/l	Ba mg/l	Cd mg/l	Cr mg/l	Cu mg/l	Hg mg/l	Mo mg/l	Ni mg/l
3.97	0.053	1.5	0.45	0.053	18	0.00062	0.0028	0.86
5.99	0.047	1.1	0.12	0.047	0.099	0.00088	0.0978	0.17
6.91	0.053	0.58	0.012	0.068	0.012	0.0019	0.12	0.036
7.96	0.041	0.59	0.0025	0.099	0.0041	0.0019	0.10	0.013
8.93	0.034	0.60	0.00045	0.094	0.0025	0.0016	0.073	0.0089
10.00	0.033	0.33	0.00014	0.11	0.0027	0.0015	0.043	0.0061
10.70	0.017	0.18	0.000069	0.049	0.0034	0.0019	0.028	0.0030
12.01	0.013	0.11	0.00011	0.041	0.047	0.0019	0.043	0.000167

pH	Pb mg/l	Sb mg/l	Se mg/l	Zn mg/l	Chloride mg/l	Fluoride mg/l	Sulphate mg/l	DOC mg/l
3.97	3.3	0.032	0.043	189	1790		1211	
5.99	0.042	0.11	0.058	12	1874		1158	
6.91	0.0056	0.20	0.077	0.51	2023		1075	
7.96	0.0015	0.26	0.059	0.07	1967		1083	
8.93	0.00016	0.067	0.032	0.2	1747		1176	
10.00	0.00012	0.0050	0.022	0.04	1793		1957	
10.70	0.0011	0.0015	0.031	0.0005	1920		1264	
12.01	0.053	0.015	0.027	0.02	1990		7574	







Experimental test of tank leaching procedure (CEN/TC 292/WG6)

DHI/2004

Material: BCR-1, cement stabilised MSWI fly ash

DHI-ID: R-233-04A

Side a:	4.0	cm	Area:	92.82	cm ²
Side b:	3.9	cm	Volume:	60.84	cm ³
Side c:	3.9	cm	Weight:	132.94	g

Fraction	Contact time days	Accum. time days	Volume ml	L/A ml/cm ²	pH	Conductivity mS/m
E1	0.25	0.25	186	2.0	11.21	117
E2	3.0	3.25	186	2.0	11.70	240
E3-1	2	5.25	742	8.0	11.01	51
E3-2	5	10.25	741	8.0	11.02	66
E3-3	7	17.25	744	8.0	11.05	61
E3-4	1	18.25	740	8.0	10.68	19
E3-5	19	37.25	751	8.1	11.36	120
E3-6	28	65.25	743	8.0	11.57	106

Measured concentrations in eluates:

Fraction	As mg/l	Ba mg/l	Cd mg/l	Cr mg/l	Cu mg/l	Hg mg/l	Mo mg/l	Ni mg/l
E1	0.0043	0.021	0.000027	0.065	0.0010	0.0016	0.0065	0.000078
E2	0.0033	0.026	0.000032	0.092	0.0028	0.0050	0.0081	0.000065
E3-1	0.0020	0.025	0.000014	0.017	0.00068	0.0023	0.00083	0.000013
E3-2	0.0291	0.04	0.000015	0.011	0.00085	0.0019	0.00091	0.00004
E3-3	0.0021	0.024	0.000009	0.017	0.00046	0.0017	0.00056	0.000015
E3-4	0.0016	0.021	0.000011	0.012	0.00029	0.0016	0.00017	0.000055
E3-5	0.0086	0.25	0.00010	0.099	0.00070	0.00010	0.00018	0.000087
E3-6	0.0014	0.043	0.00026	0.015	0.0010	0.00017	0.00050	0.000001

Fraction	Pb mg/l	Sb mg/l	Se mg/l	Zn mg/l	Chloride mg/l	Fluoride mg/l	Sulphate mg/l	DOC mg/l
E1	0.0012	0.0052	0.0061	0.002	71		80	
E2	0.0019	0.0109	0.0074	0.002	230		69	
E3-1	0.00095	0.0022	0.00025	0.0005	23		3.1	
E3-2	0.00018	0.000024	0.00043	0.002	40		3.7	
E3-3	0.00090	0.0026	0.00060	0.001	41		2.7	
E3-4	0.00042	0.00096	0.00062	0.001	7.1		0.05	
E3-5	0.0013	0.00091	0.00084	0.04	60		3.8	
E3-6	0.0052	0.00357	0.00063	0.01	207		1.4	

Experimental test of tank leaching procedure (CEN/TC 292/WG6)

DHI/2004

Material: BCR-1, cement stabilised MSWI fly ash

DHI-ID: R-233-04A

Side a:	4	cm	Area:	92.82	cm ²
Side b:	3.9	cm	Volume:	60.84	cm ³
Side c:	3.9	cm	Weight:	132.94	g

Fraction	Contact time days	Accum. time days	Volume ml	L/A ml/cm ²	pH	Conductivity mS/m
E1	0.25	0.25	186	2.0	11.21	117
E2	3.0	3.25	186	2.0	11.7	240
E3-1	2	5.25	742	8.0	11.01	51
E3-2	5	10.25	741	8.0	11.02	66
E3-3	7	17.25	744	8.0	11.05	61
E3-4	1	18.25	740	8.0	10.68	19
E3-5	19	37.25	751	8.1	11.36	120
E3-6	28	65.25	743	8.0	11.57	106

Calculated release during each step:

Fraction	As mg/m ²	Ba mg/m ²	Cd mg/m ²	Cr mg/m ²	Cu mg/m ²	Hg mg/m ²	Mo mg/m ²	Ni mg/m ²
E1	0.086	0.42	0.00054	1.297	0.0207	0.0327	0.1303	0.00156
E2	0.066	0.52	0.00064	1.851	0.0566	0.1002	0.1623	0.00130
E3-1	0.162	2.00	0.00112	1.326	0.0545	0.1875	0.0663	0.00104
E3-2	2.321	3.19	0.00120	0.913	0.0682	0.1508	0.0726	0.00319
E3-3	0.168	1.92	0.00072	1.347	0.0369	0.1324	0.0449	0.00120
E3-4	0.130	1.67	0.00088	0.917	0.0230	0.1236	0.0136	0.00438
E3-5	0.694	20.23	0.00809	8.030	0.0570	0.0079	0.0146	0.00704
E3-6	0.112	3.44	0.02081	1.175	0.0813	0.0134	0.0400	0.00008

Fraction	Pb mg/m ²	Sb mg/m ²	Se mg/m ²	Zn mg/m ²	Chloride mg/m ²	Fluoride mg/m ²	Sulphate mg/m ²	DOC mg/m ²
E1	0.0247	0.104	0.121	0.04008	1423		1603	
E2	0.0390	0.218	0.148	0.04008	4609		1383	
E3-1	0.0762	0.176	0.020	0.03997	1839		248	
E3-2	0.0146	0.002	0.034	0.15966	3193		295	
E3-3	0.0717	0.206	0.048	0.08016	3286		216	
E3-4	0.0336	0.076	0.050	0.07972	566		4	
E3-5	0.1040	0.074	0.068	3.23637	4855		307	
E3-6	0.4178	0.286	0.051	0.80047	16570		112	

Experimental test of tank leaching procedure (CEN/TC 292/WG6)

DHI/2004

Material: BCR-1, cement stabilised MSWI fly ash

DHI-ID: R-233-04A

Side a:	4	cm	Area:	92.82	cm ²
Side b:	3.9	cm	Volume:	60.84	cm ³
Side c:	3.9	cm	Weight:	132.94	g

Fraction	Contact time days	Accum. time days	Volume ml	L/A ml/cm ²	pH	Conductivity mS/m
E1	0.25	0.25	186	2.0	11.21	117
E2	3	3.25	186	2.0	11.7	240
E3-1	2	5.25	742	8.0	11.01	51
E3-2	5	10.25	741	8.0	11.02	66
E3-3	7	17.25	744	8.0	11.05	61
E3-4	1	18.25	740	8.0	10.68	19
E3-5	19	37.25	751	8.1	11.36	120
E3-6	28	65.25	743	8.0	11.57	106

Accumulated release:

Fraction	As mg/m ²	Ba mg/m ²	Cd mg/m ²	Cr mg/m ²	Cu mg/m ²	Hg mg/m ²	Mo mg/m ²	Ni mg/m ²
E1	0.0857259	0.420814	0.000541	1.29707	0.02068	0.032743374	0.130252	0.001563
E2	0.1519541	0.941823	0.001182	3.148334	0.07725	0.132917259	0.292566	0.002866
E3-1	0.3141517	2.940315	0.002301	4.474293	0.131768	0.320375781	0.358916	0.003905
E3-2	2.6351055	6.133592	0.003499	5.386852	0.199945	0.471178302	0.431563	0.007098
E3-3	2.8035114	8.057315	0.00422	6.733779	0.236816	0.603594592	0.47645	0.0083
E3-4	2.9338605	9.731523	0.005097	7.651165	0.259856	0.727167098	0.490003	0.012685
E3-5	3.6274149	29.95885	0.013188	15.68076	0.316817	0.735096208	0.504567	0.019724
E3-6	3.7393212	33.40088	0.034001	16.85602	0.398145	0.748464124	0.544591	0.019804

Fraction	Pb mg/m ²	Sb mg/m ²	Se mg/m ²	Zn mg/m ²	Chloride mg/m ²	Fluoride mg/m ²	Sulphate mg/m ²	DOC mg/m ²
E1	0.0247479	0.104222	0.121495	0.040078	1422.754		1603.103	
E2	0.0637634	0.322624	0.269201	0.080155	6031.674		2985.779	
E3-1	0.1399459	0.498572	0.289426	0.120125	7870.287		3233.592	
E3-2	0.1545552	0.500488	0.323754	0.279789	11063.56		3528.97	
E3-3	0.226294	0.706807	0.371606	0.359944	14349.92		3745.389	
E3-4	0.2598579	0.783103	0.421195	0.439668	14915.97		3749.375	
E3-5	0.3638263	0.856811	0.489401	3.67604	19770.52		4056.83	
E3-6	0.7816738	1.142581	0.540071	4.476514	36340.34		4168.897	

Experimental test of tank leaching procedure (CEN/TC 292/WG6)**Material:** BCR-1, cement stabilised MSWI fly ash**DHI-ID:** R-233-04A

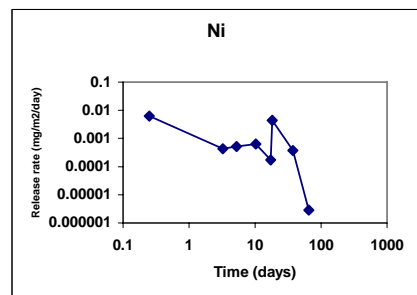
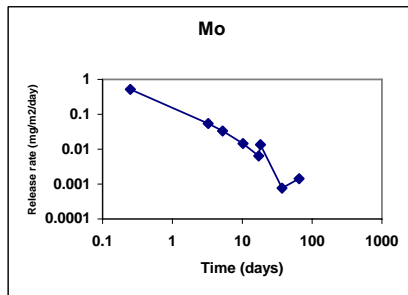
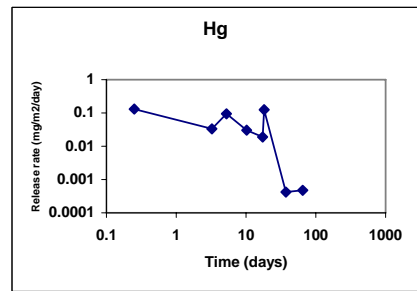
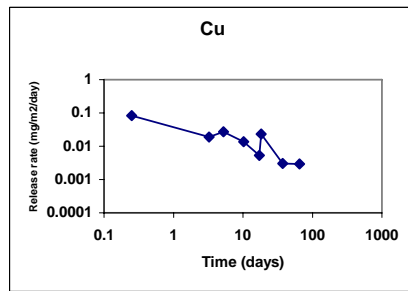
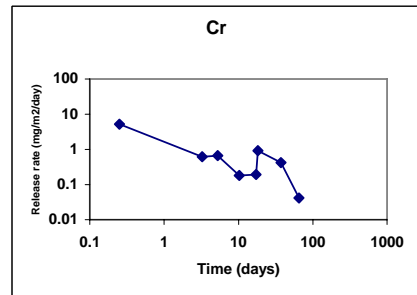
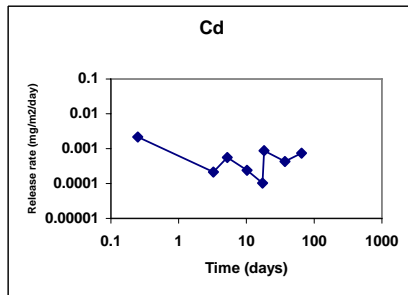
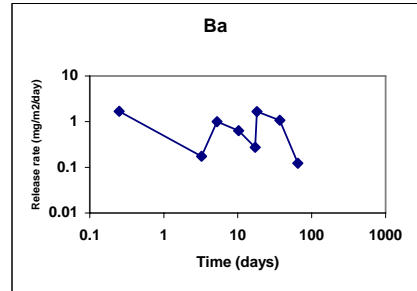
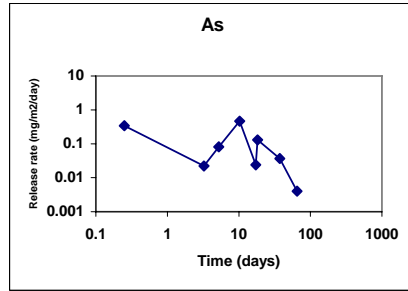
Side a:	4	cm	Area:	92.82	cm ²
Side b:	3.9	cm	Volume:	60.84	cm ³
Side c:	3.9	cm	Weight:	132.94	g

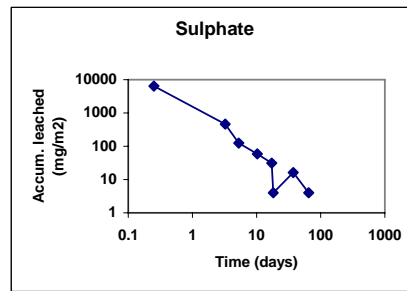
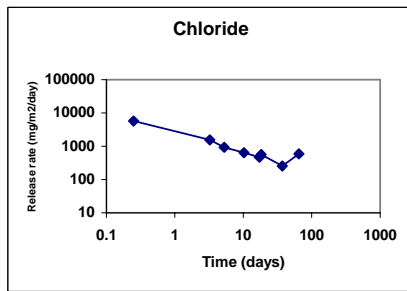
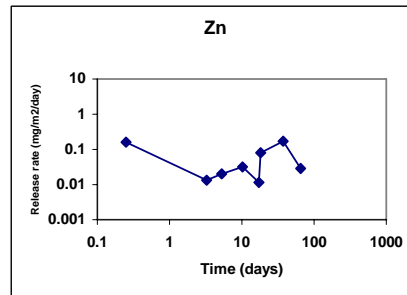
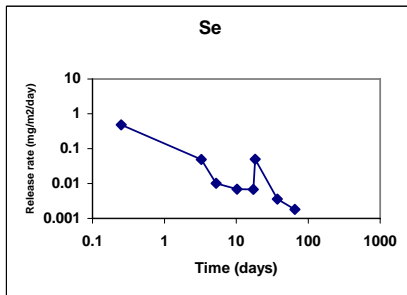
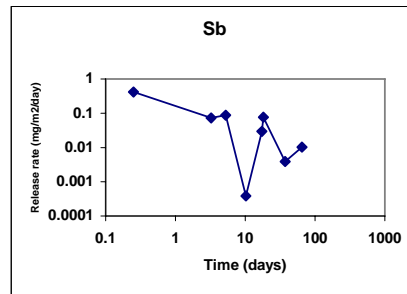
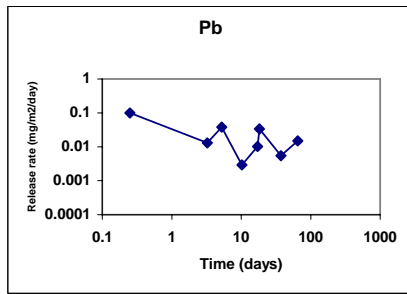
Fraction	Contact time days	Accum. time days	Volume ml	L/A ml/cm ²	pH	Conductivity mS/m
E1	0.25	0.25	186	2.0	11.21	117
E2	3	3.25	186	2.0	11.7	240
E3-1	2	5.25	742	8.0	11.01	51
E3-2	5	10.25	741	8.0	11.02	66
E3-3	7	17.25	744	8.0	11.05	61
E3-4	1	18.25	740	8.0	10.68	19
E3-5	19	37.25	751	8.1	11.36	120
E3-6	28	65.25	743	8.0	11.57	106

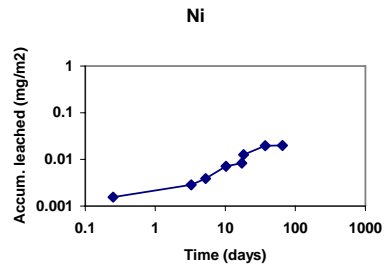
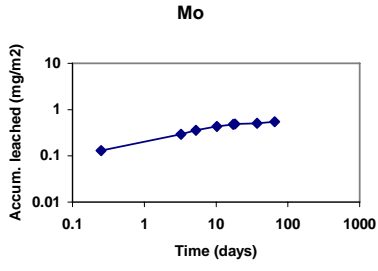
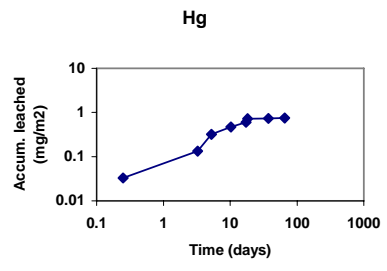
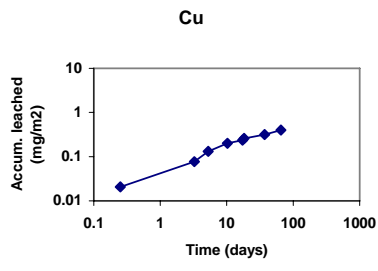
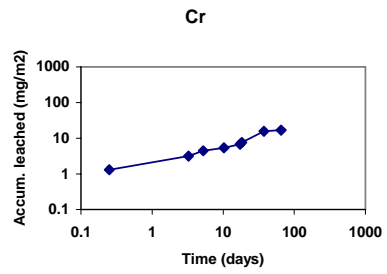
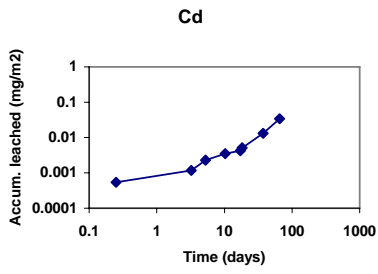
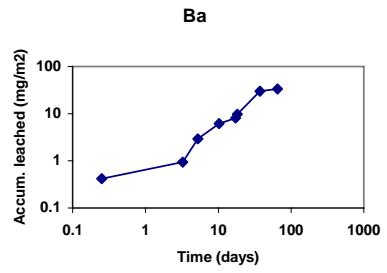
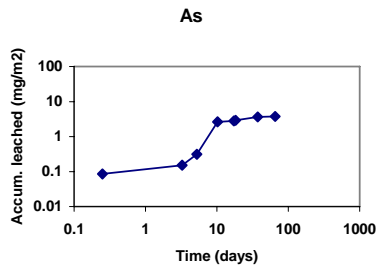
Release rates for each fraction:

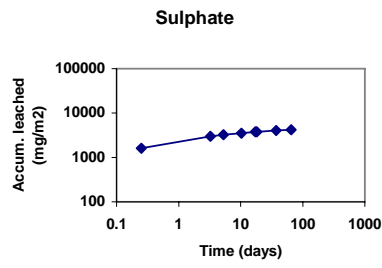
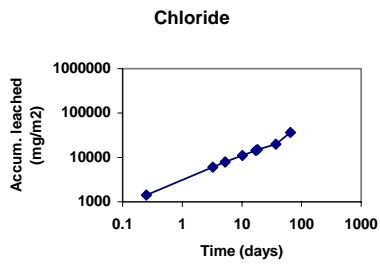
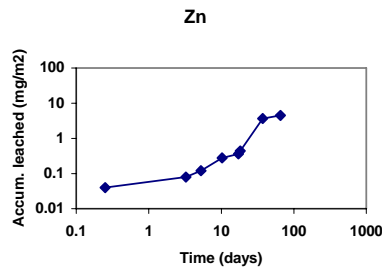
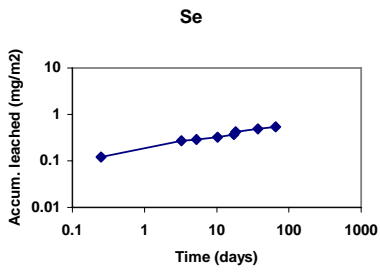
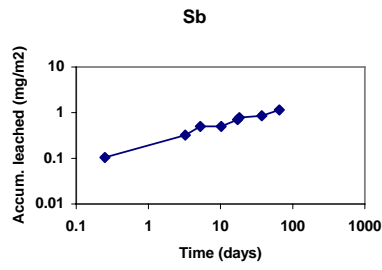
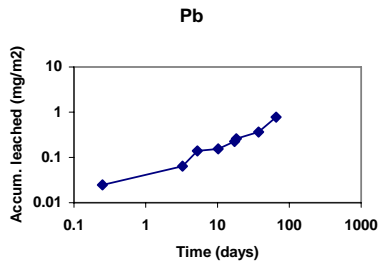
Fraction	As mg/m ² /d	Ba mg/m ² /d	Cd mg/m ² /d	Cr mg/m ² /d	Cu mg/m ² /d	Hg mg/m ² /d	Mo mg/m ² /d	Ni mg/m ² /d
E1	0.3429037	1.683258	0.002164	5.188282	0.08272	0.130973497	0.521008	0.006252
E2	0.0220761	0.173669	0.000214	0.617088	0.018856	0.033391295	0.054105	0.000434
E3-1	0.0810988	0.999246	0.00056	0.66298	0.027259	0.093729261	0.033175	0.00052
E3-2	0.4641908	0.638655	0.000239	0.182512	0.013635	0.030160504	0.014529	0.000639
E3-3	0.024058	0.274818	0.000103	0.192418	0.005267	0.018916613	0.006412	0.000172
E3-4	0.1303491	1.674208	0.000877	0.917386	0.02304	0.123572506	0.013553	0.004385
E3-5	0.0365029	1.064596	0.000426	0.42261	0.002998	0.000417322	0.000767	0.00037
E3-6	0.0039967	0.12293	0.000743	0.041973	0.002905	0.000477426	0.001429	2.86E-06

Fraction	Pb mg/m ² /d	Sb mg/m ² /d	Se mg/m ² /d	Zn mg/m ² /d	Chloride mg/m ² /d	Fluoride mg/m ² /d	Sulphate mg/m ² /d	DOC mg/m ² /d
E1	0.0989916	0.416887	0.485981	0.16031	5691		6412	
E2	0.0130052	0.072801	0.049235	0.013359	1536		461	
E3-1	0.0380913	0.087974	0.010112	0.019985	919		124	
E3-2	0.0029218	0.000383	0.006866	0.031933	639		59	
E3-3	0.0102484	0.029474	0.006836	0.011451	469		31	
E3-4	0.0335639	0.076296	0.049588	0.079724	566		4	
E3-5	0.005472	0.003879	0.00359	0.170335	256		16	
E3-6	0.0149231	0.010206	0.00181	0.028588	592		4	









**pH-dependent leaching data provided by ECN
(ANC test performed at L/S = 10 l/kg)**

Material: BCR-1, cement stabilised MSWI fly ash

Measured concentrations in eluates:

pH	Ba mg/l	Cd mg/l	Cr mg/l	Cu mg/l	Mo mg/l
4.08	2.42	2.62	0.246	4.74	0.0174
5.52	1.94	2.26	0.134	1.19	0.0414
7.50	1.64	0.292	0.711	0.0484	0.103
8.78	1.58	0.0549	0.857	0.0102	0.098
9.96	1.11	0.0028	0.917	0.004	0.103
11.00	1.10	0.0011	0.798	0.004	0.101
11.61	7.35	0.0006	0.160	0.0045	0.0558
12.55	3.38	0.0007	0.0655	0.0033	0.0218

pH	Ni mg/l	Pb mg/l	Sb mg/l	SO4- mg/l	Zn mg/l
4.08	9.92	13.0	0.0062	825	140
5.52	4.17	0.742	0.0355	799	118
7.50	1.49	0.110	0.0806	814	2.28
8.78	0.209	0.0112	0.075	844	1.26
9.96	0.0029	0.0396	0.0736	1241	0.084
11.00	0.0029	0.0141		1138	0.0415
11.61	0.0029	0.0814	0.0069	20.9	0.0673
12.55	0.0029	0.263		15.8	0.104

Leached amounts:

pH	Ba mg/kg	Cd mg/kg	Cr mg/kg	Cu mg/kg	Mo mg/kg
4.08	24.24	26.25	2.46	47.40	0.17
5.52	19.44	22.63	1.34	11.92	0.41
7.50	16.40	2.92	7.11	0.48	1.03
8.78	15.82	0.55	8.57	0.10	0.98
9.96	11.11	0.028	9.17	0.040	1.03
11.00	10.96	0.011	7.98	0.040	1.01
11.61	73.52	0.006	1.60	0.045	0.56
12.55	33.77	0.007	0.66	0.033	0.22

pH	Ni mg/kg	Pb mg/kg	Sb mg/kg	SO4- mg/kg	Zn mg/kg
4.08	99.21	129.91	0.06	8249	1399
5.52	41.71	7.42	0.36	7991	1178
7.50	14.90	1.10	0.81	8140	22.83
8.78	2.09	0.11	0.75	8438	12.55
9.96	0.029	0.40	0.74	12411	0.84
11.00	0.029	0.14		11383	0.42
11.61	0.029	0.81	0.069	209	0.67
12.55	0.029	2.63		158	1.04

Tank leaching data provided by ECN (NEN 7345)

2005 april

Material: BCR-2, cement stabilised MSWI fly ash

Surface area: 553 cm²

Fraction	Contact time days	Accum. time days	Volume ml	L/A ml/cm ²	pH
1	0.25	0.25	4000	7.2	10.98
2	0.75	1	4000	7.2	11.20
3	1.25	2.25	4000	7.2	11.32
4	1.75	4	4000	7.2	11.39
5	5.00	9	4000	7.2	11.58
6	7.00	16	4000	7.2	11.56
7	20.00	36	4000	7.2	11.64
8	28.00	64	4000	7.2	11.50

Measured concentrations in eluates:

Fraction	As mg/l	Ba mg/l	Cd mg/l	Cr mg/l	Cu mg/l	Hg mg/l	Mo mg/l	Ni mg/l
1		0.085	0.00286	0.136	0.0027		0.335	0.00665
2		0.0809	0.0002	0.081	0.0027		0.222	0.00665
3		0.104	0.00013	0.073	0.00079		0.199	0.00665
4		0.125	0.00035	0.061	0.00010		0.173	0.00665
5		0.209	0.0007	0.081	0.00270		0.276	0.00665
6		0.244	0.0007	0.044	0.00540		0.204	0.00665
7		0.281	0.00070	0.050	0.00400		0.315	0.002
8		0.215	0.0007	0.039	0.0010		0.222	0.00665

Fraction	Pb mg/l	Sb mg/l	Se mg/l	Zn mg/l	Chloride mg/l	Fluoride mg/l	Sulphate mg/l	DOC mg/l
1	0.0451	0.0207		0.0316	2621		581	
2	0.0532	0.0167		0.0434	1690		317	
3	0.04646	0.0206		0.0596	1505		239	
4	0.06609	0.0193		0.0699	1217		169	
5	0.04200	0.0340		0.083	1688		166	
6	0.06700	0.0410		0.073	1164		74.9	
7	0.0710	0.0580		0.054	1504		69.5	
8	0.0430	0.0560		0.036	781		45.8	

Tank leaching data provided by ECN (NEN 7345)**Material:** BCR-2, cement stabilised MSWI fly ash**Surface area:** 553 cm²

Fraction	Contact time days	Accum. time days	Volume ml	L/A ml/cm ²	pH
1	0.25	0.25	4000	7.2	10.98
2	0.75	1	4000	7.2	11.2
3	1.25	2.25	4000	7.2	11.32
4	1.75	4	4000	7.2	11.39
5	5.00	9	4000	7.2	11.58
6	7.00	16	4000	7.2	11.56
7	20.00	36	4000	7.2	11.64
8	28.00	64	4000	7.2	11.5

Calculated release during each step:

Fraction	As mg/m ²	Ba mg/m ²	Cd mg/m ²	Cr mg/m ²	Cu mg/m ²	Hg mg/m ²	Mo mg/m ²	Ni mg/m ²
1		6.15	0.20687	9.825	0.1953		24.2271	0.48101
2		5.85	0.01447	5.878	0.1953		16.0680	0.48101
3		7.49	0.00940	5.315	0.0571		14.3747	0.48101
4		9.06	0.02532	4.434	0.0072		12.4926	0.48101
5		15.12	0.05063	5.859	0.1953		19.9638	0.48101
6		17.65	0.05063	3.183	0.3906		14.7559	0.48101
7		20.33	0.05063	3.617	0.2893		22.7848	0.14467
8		15.55	0.05063	2.821	0.0723		16.0579	0.48101

Fraction	Pb mg/m ²	Sb mg/m ²	Se mg/m ²	Zn mg/m ²	Chloride mg/m ²	Fluoride mg/m ²	Sulphate mg/m ²	DOC mg/m ²
1	3.2600	1.499		2.28788	189584		41992	
2	3.8452	1.209		3.13707	122242		22930	
3	3.3606	1.493		4.31031	108861		17306	
4	4.7805	1.395		5.05316	88029		12244	
5	3.0380	2.459		6.00362	122098		12002	
6	4.8463	2.966		5.28029	84195		5414	
7	5.1356	4.195		3.90597	108788		5026	
8	3.1103	4.051		2.60398	56492		3310	

Tank leaching data provided by ECN (NEN 7345)**Material:** BCR-2, cement stabilised MSWI fly ash**Surface area:** 553 cm²

Fraction	Contact time days	Accum. time days	Volume ml	L/A ml/cm ²	pH
1	0.25	0.25	4000	7.2	10.98
2	0.75	1	4000	7.2	11.2
3	1.25	2.25	4000	7.2	11.32
4	1.75	4	4000	7.2	11.39
5	5.00	9	4000	7.2	11.58
6	7.00	16	4000	7.2	11.56
7	20.00	36	4000	7.2	11.64
8	28.00	64	4000	7.2	11.5

Accumulated release:

Fraction	As mg/m ²	Ba mg/m ²	Cd mg/m ²	Cr mg/m ²	Cu mg/m ²	Hg mg/m ²	Mo mg/m ²	Ni mg/m ²
1		6.146835	0.206872	9.824955	0.195298		24.22712	0.481013
2		12.00072	0.221338	15.70271	0.390597		40.29512	0.962025
3		19.48716	0.230741	21.01772	0.44774		54.6698	1.443038
4		28.54828	0.256058	25.45172	0.454973		67.16239	1.924051
5		43.66582	0.306691	31.31067	0.650271		87.12622	2.405063
6		61.31501	0.357324	34.49331	1.040868		101.8821	2.886076
7		81.64051	0.407957	38.10995	1.330199		124.6669	3.030741
8		97.19204	0.45859	40.93092	1.402532		140.7248	3.511754

Fraction	Pb mg/m ²	Sb mg/m ²	Se mg/m ²	Zn mg/m ²	Chloride mg/m ²	Fluoride mg/m ²	Sulphate mg/m ²	DOC mg/m ²
1	3.2600362	1.499458		2.287884	189584.1		41992.26	
2	7.1052441	2.708861		5.424955	311826.4		64922.61	
3	10.465823	4.201808		9.735262	420687.2		82228.66	
4	15.246293	5.596383		14.78843	508716.1		94472.44	
5	18.284268	8.055696		20.79204	630813.7		106474.8	
6	23.130561	11.02134		26.07233	715009		111888.9	
7	28.266184	15.21664		29.9783	823797.5		116915.3	
8	31.376492	19.26727		32.58228	880289.3		120225.4	

Tank leaching data provided by ECN (NEN 7345)

Material: BCR-2, cement stabilised MSWI fly ash

Surface area: 553 cm²

Fraction	Contact time days	Accum. time days	Volume ml	L/A ml/cm ²	pH
1	0.25	0.25	4000	7.2	10.98
2	0.75	1	4000	7.2	11.2
3	1.25	2.25	4000	7.2	11.32
4	1.75	4	4000	7.2	11.39
5	5.00	9	4000	7.2	11.58
6	7.00	16	4000	7.2	11.56
7	20.00	36	4000	7.2	11.64
8	28.00	64	4000	7.2	11.5

Release rates for each fraction:

Fraction	As mg/m ² /d	Ba mg/m ² /d	Cd mg/m ² /d	Cr mg/m ² /d	Cu mg/m ² /d	Hg mg/m ² /d	Mo mg/m ² /d	Ni mg/m ² /d
1		24.58734	0.827486	39.29982	0.781193		96.9085	1.924051
2		7.805184	0.019289	7.83701	0.260398		21.42399	0.64135
3		5.98915	0.007523	4.252007	0.045714		11.49975	0.38481
4		5.177784	0.014467	2.533712	0.004133		7.138621	0.274864
5		3.023508	0.010127	1.17179	0.03906		3.992767	0.096203
6		2.521312	0.007233	0.454663	0.0558		2.107982	0.068716
7		1.016275	0.002532	0.180832	0.014467		1.139241	0.007233
8		0.555412	0.001808	0.100749	0.002583		0.573495	0.017179

Fraction	Pb mg/m ² /d	Sb mg/m ² /d	Se mg/m ² /d	Zn mg/m ² /d	Chloride mg/m ² /d	Fluoride mg/m ² /d	Sulphate mg/m ² /d	DOC mg/m ² /d
1	13.040145	5.99783		9.151537	758336		167969	
2	5.1269439	1.612538		4.182761	162990		30574	
3	2.6884629	1.194358		3.448246	87089		13845	
4	2.7316972	0.7969		2.887523	50302		6996	
5	0.6075949	0.491863		1.200723	24420		2400	
6	0.6923276	0.423663		0.754327	12028		773	
7	0.2567812	0.209765		0.195298	5439		251	
8	0.1110824	0.144665		0.092999	2018		118	

**pH-dependent leaching data provided by ECN
(test performed at L/S = 10 l/kg)**

Material: BCR-2, cement stabilised MSWI fly ash

Measured concentrations in eluates:

pH	Ba mg/l	Cd mg/l	Cr mg/l	Cu mg/l	Mo mg/l	Ni mg/l
4	1.27	17.8	0.969	36.5	0.013	0.929
5	1.77	16.6	0.031	18.4	0.015	0.807
6	1.92	14.3	0.006	2.20	0.14	0.639
7	1.84	8.72	0.116	0.029	0.275	0.388
8	1.93	1.78	0.115	0.007	0.132	0.161
10	1.11	0.035	0.205	0.005	0.173	0.003
10	1.17	0.002	0.414	0.0015	0.443	0.003
12	0.18	0.003	0.134	0.016	0.77	0.003

pH	Pb mg/l	Sb mg/l	Zn mg/l	Cl- mg/l	SO4- mg/l
4	95.5	0.222	801	4725	1285
5	41.1	0.114	721	4676	1087
6	4.29	0.304	602	4809	1105
7	0.269	0.332	61.9	4816	1065
8	0.104	0.311	5.67	5350	1132
10	0.023	0.492	0.098	4775	1184
10	0.028	0.352	0.124	4906	1132
12	1.86	0.009	1.35	4919	3197

Experimental test of tank leaching procedure (CEN/TC 292/WG6)

VTT/2004

Material: BCR-1, cement stabilised MSWI fly ash
VTT-Code: Sample 1

Side a: 4.0 cm **Area:** 92.02 cm²
Side b: 3.9 cm **Volume:** 60.05 cm³
Side c: 3.8 cm **Weight:** 129.90 g

Fraction	Contact time days	Accum. time days	Volume l	L/A ml/cm ²	pH	Cond. mS/m
E1	0.26	0.26	0.18441	2.0	11.6	192
E2	2.00	2.26	0.18428	2.0	11.6	199
E3-1	2.22	4.23	0.736	8.0	11.4	55
E3-2	7.02	9.24	0.73519	8.0	11.4	73
E3-3	9.23	16.25	0.73559	8.0	11.3	71
E3-4	8.02	17.25	0.73619	8.0	10.9	16
E3-5	28.23	36.25	0.73519	8.0	11.4	109
E3-6	36.04	64.27	0.74329	8.1	11.1	111

Measured concentrations in eluates:

Fraction	As mg/l	Ba mg/l	Cd mg/l	Cr mg/l	Cu mg/l	Mn mg/l	Mo mg/l	Ni mg/l
E1	0.012	0.0114	<0.00003	0.0819	0.0032	0.0007	0.0107	0.0005
E2	0.0006	0.0189	<0.00003	0.0296	0.001	0.0003	0.0043	0.0004
E3-1	0.001	0.0185	<0.00002	0.004	0.0035	0.0004	0.0005	0.0003
E3-2	0.0006	0.0308	<0.00002	0.0047	0.003	0.0004	0.0006	0.0003
E3-3	0.0005	0.0334	<0.00002	0.0038	0.002	0.0003	0.0005	0.0005
E3-4	0.0003	0.0103	<0.00002	0.0014	0.0032	0.0003	0.0001	0.0003
E3-5	0.00064	0.0515	<0.0002	0.00339	<0.001	0.0003	0.00104	0.00088
E3-6	0.00062	0.0611	<0.0002	0.00344	0.0014	0.0005	0.00075	0.00397

Fraction	Pb mg/l	Sb mg/l	Se mg/l	Zn mg/l	Chloride mg/l	Fluoride mg/l	Sulphate mg/l	Al mg/l	Na mg/l	Ca mg/l	K mg/l
E1	0.0027	0.0134	0.0098	0.0343	109	3.6	131	1.28	165	18.3	212
E2	0.0022	0.0141	<0.005	0.0068	134	3.8	26.2	2.85	133.0	27.7	197
E3-1	0.0016	0.0033	<0.0005	0.0055	21.7	1.2	2.3	1.05	16.8	34.5	31.3
E3-2	0.0025	0.0042	0.0006	0.0066	42.1	1.4	2.7	1.49	28.6	42.9	54.5
E3-3	0.0024	0.0043	0.001	0.0061	45.3	1.4	2.6	1.62	29	42.1	56.9
E3-4	0.0008	0.0017	<0.0005	0.0041	5.7	0.4	1	0.62	3.99	16.1	8.01
E3-5	0.0036	<0.005	0.0046	0.01	92.8	<1	3.4	2.21	51	46.2	81.3
E3-6	0.00374	<0.005	0.00471	0.01	82.5	<1	3.1	2.280	47.4	44.5	73.4

Experimental test of tank leaching procedure (CEN/TC 292/WG6)

VTT/2004

Material: BCR-1, cement stabilised MSWI fly ash
VTT-Code: Sample 1

Side a: 4.01 cm **Area:** 92.02 cm²
Side b: 3.92 cm **Volume:** 60.05 cm³
Side c: 3.82 cm **Weight:** 129.90 g

Fraction	Contact time days	Accum. time days	Volume l	L/A ml/cm ²	pH	Cond. mS/m
E1	0.26	0.26	0.18441	2.0	11.6	192
E2	2.00	2.26	0.18428	2.0	11.6	199
E3-1	2.22	4.23	0.736	8.0	11.4	55
E3-2	7.02	9.24	0.73519	8.0	11.4	73
E3-3	9.23	16.25	0.73559	8.0	11.3	71
E3-4	8.02	17.25	0.73619	8.0	10.9	16
E3-5	28.23	36.25	0.73519	8.0	11.4	109
E3-6	36.04	64.27	0.74329	8.1	11.1	111

Calculated release during each step:

Fraction	As mg/m ²	Ba mg/m ²	Cd mg/m ²	Cr mg/m ²	Cu mg/m ²	Mn mg/m ²	Mo mg/m ²	Ni mg/m ²
E1	0.236	0.23		1.64	0.06	0.01	0.21	0.01
E2	0.012	0.38		0.59	0.02	0.01	0.09	0.01
E3-1	0.040	1.48		0.32	0.28	0.03	0.04	0.02
E3-2	0.048	2.46		0.38	0.24	0.03	0.05	0.02
E3-3	0.040	2.67		0.30	0.16	0.02	0.04	0.04
E3-4	0.024	0.82		0.11	0.26	0.02	0.01	0.02
E3-5	0.051	4.11		0.27	0.08	0.02	0.08	0.07
E3-6	0.050	4.94		0.28	0.11	0.04	0.06	0.32

Fraction	Pb mg/m ²	Sb mg/m ²	Se mg/m ²	Zn mg/m ²	Chloride mg/m ²	Fluoride mg/m ²	Sulphate mg/m ²	Al mg/m ²	Na mg/m ²	Ca mg/m ²	K mg/m ²
E1	0.05	0.269		0.69	2184	72.1	2625	25.7	3307	367	4248
E2	0.04	0.282		0.14	2683	76.1	525	57.1	2663	555	3945
E3-1	0.13	0.264		0.44	1760	96.0	184	84.0	1344	2759	2503
E3-2	0.20	0.336		0.53	3355	111.8	216	119.0	2285	3427	4354
E3-3	0.19	0.344		0.49	3597	111.9	208	129.5	2318	3365	4548
E3-4	0.06	0.136		0.33	456	32.0	80	49.6	319	1288	641
E3-5	0.29	0.399		0.80	7430	79.9	272	176.6	4074	3691	6495
E3-6	0.30	0.404		0.81	6704	80.8	250	184.2	3829	3594	5929

Experimental test of tank leaching procedure (CEN/TC 292/WG6)

VTT/2004

Material: BCR-1, cement stabilised MSWI fly ash
VTT-Code: Sample 1

Side a: 4.01 cm **Area:** 92.02 cm²
Side b: 3.92 cm **Volume:** 60.05 cm³
Side c: 3.82 cm **Weight:** 129.90 g

Accumulated release:

Fraction	As mg/m ²	Ba mg/m ²	Cd mg/m ²	Cr mg/m ²	Cu mg/m ²	Mn mg/m ²	Mo mg/m ²	Ni mg/m ²
E1	0.236	0.228		1.641	0.064	0.014	0.214	0.010
E2	0.248	0.607		2.234	0.084	0.020	0.301	0.018
E3-1	0.288	2.087		2.554	0.364	0.052	0.341	0.042
E3-2	0.336	4.547		2.929	0.604	0.084	0.388	0.066
E3-3	0.376	7.217		3.233	0.764	0.108	0.428	0.106
E3-4	0.400	8.041		3.345	1.020	0.132	0.436	0.130
E3-5	0.452	12.155		3.616	1.100	0.154	0.520	0.200
E3-6	0.502	17.091		3.894	1.213	0.195	0.580	0.521

Fraction	Pb mg/m ²	Sb mg/m ²	Se mg/m ²	Zn mg/m ²	Chloride mg/m ²	Fluoride mg/m ²	Sulphate mg/m ²	Al mg/m ²	Na mg/m ²	Ca mg/m ²	K mg/m ²
E1	0.054	0.269		0.687	2184	72	2625	26	3307	367	4248
E2	0.098	0.551		0.824	4868	148	3150	83	5970	921	8193
E3-1	0.226	0.815		1.263	6627	244	3334	167	7314	3681	10697
E3-2	0.426	1.150		1.791	9983	356	3549	286	9598	7108	15051
E3-3	0.618	1.494		2.278	13580	468	3757	415	11917	10473	19599
E3-4	0.682	1.630		2.606	14036	500	3837	465	12236	11761	20240
E3-5	0.969	2.030		3.405	21466	580	4109	641	16310	15452	26735
E3-6	1.271	2.433		4.213	28170	661	4359	826	20139	19047	32664

Experimental test of tank leaching procedure (CEN/TC 292/WG6)

VTT/2004

Material: BCR-1, cement stabilised MSWI fly ash
 VTT-Code: Sample 2

Side a: 3.8 cm Area: 94.53 cm²
 Side b: 4.1 cm Volume: 62.50 cm³
 Side c: 4.0 cm Weight: 131.87 g

Fraction	Contact time days	Accum. time days	Volume ml	L/A ml/cm ²	pH	Cond. mS/m
E1	0.26	0.26	189.18	48.3	11.6	203
E2	0.98	1.24	189.52	48.3	11.6	170
E3-1	3.25	4.23	757	193.1	11.6	102
E3-2	6.00	9.25	756.53	193.0	11.5	101
E3-3	10.25	16.25	756.53	193.0	11.4	95

Measured concentrations in eluates:

Fraction	As mg/l	Ba mg/l	Cd mg/l	Cr mg/l	Cu mg/l	Mn mg/l	Mo mg/l	Ni mg/l
E1	0.006	0.0114	<0.00003	0.0793	0.0028	0.0004	0.0098	0.001
E2	0.0005	0.0189	<0.00003	0.0259	0.0015	0.0005	0.0031	0.0006
E3-1	0.000	0.0185	<0.00002	0.0063	0.0018	0.0002	0.0008	0.0003
E3-2	0.0006	0.0308	<0.00002	0.0058	0.0008	0.0002	0.0006	0.0004
E3-3	0.0005	0.0334	<0.00002	0.0048	0.0012	0.0003	0.0005	0.0005

Fraction	Pb mg/l	Sb mg/l	Se mg/l	Zn mg/l	Chloride mg/l	Fluoride mg/l	Sulphate mg/l	Al mg/l	Na mg/l	Ca mg/l	K mg/l
E1	0.0041	0.0108	0.0067	0.013	120	3.7	118	1.51	152	33.3	194
E2	0.0039	0.0086	<0.005	0.0077	93.9	<1	19	2.31	83.6	58.4	133
E3-1	0.0025	0.0042	0.0007	0.0052	40.9	1.9	4.7	1.52	29	57	46.8
E3-2	0.0031	0.0043	0.0007	0.0068	46.8	1.9	3.5	1.65	31.3	53.9	53.8
E3-3	0.003	0.0044	0.0008	0.0069	50.2	1.8	3.3	1.73	30.8	49.1	54.2

Experimental test of tank leaching procedure (CEN/TC 292/WG6)

VTT/2004

Material: BCR-1, cement stabilised MSWI fly ash
VTT-Code: Sample 2

Side a: 3.82 cm **Area:** 94.53 cm²
Side b: 4.05 cm **Volume:** 62.50 cm³
Side c: 4.04 cm **Weight:** 131.87 g

Fraction	Contact time days	Accum. time days	Volume l	L/A ml/cm ²	pH	Cond. mS/m
E1	0.26	0.26	0.18441	2.0	11.6	192
E2	2.00	2.26	0.18428	2.0	11.6	199
E3-1	2.22	4.23	0.736	8.0	11.4	55
E3-2	7.02	9.24	0.73519	8.0	11.4	73
E3-3	9.23	16.25	0.73559	8.0	11.3	71

Calculated release during each step:

Fraction	As mg/m ²	Ba mg/m ²	Cd mg/m ²	Cr mg/m ²	Cu mg/m ²	Mn mg/m ²	Mo mg/m ²	Ni mg/m ²
E1	0.126	0.42		1.59	0.06	0.01	0.20	0.02
E2	0.010	0.89		0.52	0.03	0.01	0.06	0.01
E3-1	0.032	3.44		0.50	0.14	0.02	0.06	0.02
E3-2	0.048	4.15		0.46	0.06	0.02	0.05	0.03
E3-3	0.040	3.84		0.38	0.10	0.02	0.04	0.04

Fraction	Pb mg/m ²	Sb mg/m ²	Se mg/m ²	Zn mg/m ²	Chloride mg/m ²	Fluoride mg/m ²	Sulphate mg/m ²	Al mg/m ²	Na mg/m ²	Ca mg/m ²	K mg/m ²
E1	0.08	0.216		0.26	2401	74	2361	30.2	3042	666	3882
E2	0.08	0.172		0.15	1883	20	381	46.3	1676	1171	2666
E3-1	0.20	0.336		0.42	3275	152	376	121.7	2322	4565	3748
E3-2	0.25	0.344		0.54	3745	152	280	132.0	2505	4314	4306
E3-3	0.24	0.352		0.55	4017	144	264	138.5	2465	3929	4338

Experimental test of tank leaching procedure (CEN/TC 292/WG6)

VTT/2004

Material: BCR-1, cement stabilised MSWI fly ash
VTT-Code: Sample 2

Side a: 3.82 cm **Area:** 94.53 cm²
Side b: 4.05 cm **Volume:** 62.50 cm³
Side c: 4.04 cm **Weight:** 131.87 g

Accumulated release:

Fraction	As mg/m ²	Ba mg/m ²	Cd mg/m ²	Cr mg/m ²	Cu mg/m ²	Mn mg/m ²	Mo mg/m ²	Ni mg/m ²
E1	0.126	0.418		1.587	0.056	0.008	0.196	0.020
E2	0.136	1.308		2.106	0.086	0.018	0.258	0.032
E3-1	0.168	4.752		2.611	0.230	0.034	0.322	0.056
E3-2	0.216	8.905		3.075	0.294	0.050	0.370	0.088
E3-3	0.256	12.747		3.459	0.390	0.074	0.410	0.128

Fraction	Pb mg/m ²	Sb mg/m ²	Se mg/m ²	Zn mg/m ²	Chloride mg/m ²	Fluoride mg/m ²	Sulphate mg/m ²	Al mg/m ²	Na mg/m ²	Ca mg/m ²	K mg/m ²
E1	0.082	0.216		0.260	2401	74	2361	30	3042	666	3882
E2	0.160	0.389		0.415	4284	94	2742	77	4718	1837	6549
E3-1	0.360	0.725		0.831	7559	246	3119	198	7040	6402	10297
E3-2	0.609	1.069		1.375	11305	398	3399	330	9545	10715	14602
E3-3	0.849	1.421		1.927	15322	542	3663	469	12010	14645	18940

Experimental test of tank leaching procedure (CEN/TC 292/WG6)

VTT/2004

Material: BCR-1, cement stabilised MSWI fly ash
VTT-Code: Sample 3

Side a: 3.8 cm **Area:** 94.53 cm²
Side b: 4.1 cm **Volume:** 62.50 cm³
Side c: 4.0 cm **Weight:** 131.87 g

Fraction	Contact time days	Accum. time days	Volume ml	L/A ml/cm ²	pH	Cond. mS/m
E1	0.25	0.25	194	47.9	11.5	194
E2	2.00	2.25	194.41	48.0	11.7	246
E3-1	2.25	4.25	773.1	190.9	11.3	42
E3-2	7.00	9.25	773.32	190.9	11.5	93
E3-3	9.25	16.25	772.52	190.7	11.4	87

Measured concentrations in eluates:

Fraction	As mg/l	Ba mg/l	Cd mg/l	Cr mg/l	Cu mg/l	Mn mg/l	Mo mg/l	Ni mg/l
E1	0.014	0.0085	<0.00003	0.0935	0.003	0.0005	0.0101	0.0003
E2	0.0016	0.0118	<0.00003	0.0505	0.0018	0.0005	0.0051	0.0002
E3-1	0.000	0.0115	<0.00002	0.0034	0.0023	0.0002	0.0002	0.0002
E3-2	0.0008	0.0295	<0.00002	0.0059	0.0004	0.0001	0.0006	0.0004
E3-3	0.0008	0.0337	<0.00002	0.0052	0.0022	0.0002	0.0005	0.0013

Fraction	Pb mg/l	Sb mg/l	Se mg/l	Zn mg/l	Chloride mg/l	Fluoride mg/l	Sulphate mg/l	Al mg/l	Na mg/l	Ca mg/l	K mg/l
E1	0.0031	0.0131	0.0099	0.0057	109	3.7	133	1.01	171	12.9	220
E2	0.0028	0.0164	<0.005	0.005	134	4.3	28.3	3.31	180.0	12	270
E3-1	0.0009	0.0022	<0.0005	0.0107	21.7	<1	1.2	0.76	9.58	24.8	17.9
E3-2	0.0027	0.0046	0.0006	0.0077	42.1	1.9	3.1	2.08	30	49.5	48.8
E3-3	0.003	0.0051	0.0009	0.0095	45.3	1.6	2.9	2.06	29.4	47.1	54.7

Experimental test of tank leaching procedure (CEN/TC 292/WG6)

VTT/2004

Material: BCR-1, cement stabilised MSWI fly ash
VTT-Code: Sample 3

Side a: 3.82 cm **Area:** 94.53 cm²
Side b: 4.05 cm **Volume:** 62.50 cm³
Side c: 4.04 cm **Weight:** 131.87 g

Fraction	Contact time days	Accum. time days	Volume l	L/A ml/cm ²	pH	Cond. mS/m
E1	0.26	0.26	0.18441	2.0	11.6	192
E2	2.00	2.26	0.18428	2.0	11.6	199
E3-1	2.22	4.23	0.736	8.0	11.4	55
E3-2	7.02	9.24	0.73519	8.0	11.4	73
E3-3	9.23	16.25	0.73559	8.0	11.3	71

Calculated release during each step:

Fraction	As mg/m ²	Ba mg/m ²	Cd mg/m ²	Cr mg/m ²	Cu mg/m ²	Mn mg/m ²	Mo mg/m ²	Ni mg/m ²
E1	0.280	0.17		1.88	0.06	0.01	0.20	0.01
E2	0.032	0.24		1.02	0.04	0.01	0.10	0.01
E3-1	0.032	0.92		0.27	0.18	0.02	0.02	0.02
E3-2	0.064	2.36		0.47	0.03	0.01	0.05	0.02
E3-3	0.064	2.70		0.42	0.18	0.02	0.04	0.04

Fraction	Pb mg/m ²	Sb mg/m ²	Se mg/m ²	Zn mg/m ²	Chloride mg/m ²	Fluoride mg/m ²	Sulphate mg/m ²	Al mg/m ²	Na mg/m ²	Ca mg/m ²	K mg/m ²
E1	0.06	0.263		0.11	2005	74.4	2675	20.3	3439	259	4424
E2	0.06	0.330		0.10	3849	86.7	570	66.7	3627	242	5441
E3-1	0.07	0.176		0.86	874	80.1	96	60.9	768	1987	1434
E3-2	0.22	0.369		0.62	3383	152.3	248	166.7	2405	3968	3912
E3-3	0.24	0.408		0.76	3604	128.1	232	165.0	2354	3772	4380

Experimental test of tank leaching procedure (CEN/TC 292/WG6)

VTT/2004

Material: BCR-1, cement stabilised MSWI fly ash
VTT-Code: Sample 3

Side a: 3.82 cm **Area:** 94.53 cm²
Side b: 4.05 cm **Volume:** 62.50 cm³
Side c: 4.04 cm **Weight:** 131.87 g

Accumulated release:

Fraction	As mg/m ²	Ba mg/m ²	Cd mg/m ²	Cr mg/m ²	Cu mg/m ²	Mn mg/m ²	Mo mg/m ²	Ni mg/m ²
E1	0.280	0.171		1.880	0.060	0.010	0.203	0.006
E2	0.312	0.409		2.898	0.097	0.020	0.306	0.004
E3-1	0.344	1.330		3.170	0.281	0.036	0.322	0.016
E3-2	0.408	3.695		3.643	0.313	0.044	0.370	0.032
E3-3	0.472	6.394		4.060	0.489	0.060	0.410	0.104

Fraction	Pb mg/m ²	Sb mg/m ²	Se mg/m ²	Zn mg/m ²	Chloride mg/m ²	Fluoride mg/m ²	Sulphate mg/m ²	Al mg/m ²	Na mg/m ²	Ca mg/m ²	K mg/m ²
E1	0.062	0.263		0.115	2005	74	2675	20	3439	259	4424
E2	0.119	0.594		0.215	5854	161	3245	87	7066	501	9865
E3-1	0.191	0.770		1.073	6728	241	3341	148	7834	2489	11300
E3-2	0.407	1.139		1.690	10110	394	3590	315	10239	6457	15212
E3-3	0.648	1.547		2.451	13714	522	3822	480	12593	10228	19592

Appendix 3

Monolithic waste landfills in Finland

General

- in Finland only one landfill for monolithic waste material in use
- several storage fields have been constructed on landfill areas (usually cement stabilised contaminated soils)

	Landfill 1:	Landfill 2: (storage area, example)
Landfill category	Non-hazardous landfill	Not classified (1999)
Landfill area	2 ha	0,5 ha
Landfill height	5-20 m	2 m (1,5 m)
Landfill construction	"pyramide"	"rectangle"
Material	Contaminated soil	Contaminated soil
Stabilisation agent	Cement	Cement
End product requirement		
Organic material	Restriction for total content	
Metal content	Restrictions for total content	
Leaching	According to NEN 7345	NEN 7345
Compression strength	> 1 Mpa	> 3 MPA
Water permeability	< 10 ⁻⁸ m/s	10 ⁻⁹ m/s
Filling	Max. 30 cm layers, mixed and compacted at site	Mixed and compacted at site
Filling period	5 years	One summer
Ground soil	Moraine	
Barrier (bottom construction)	Asphalt (double)	Syntetic membrane (+ dranaige system)
Intermediate cover	1 m soil layer (requirements for frost resistance)	
Closing cover	Tight Asphalt	Tight asphalt (2 double layer)
Leachate treatment	Collection + monitoring	Collection + monitoring

Appendix 4

Monolithic waste landfills in Sweden, some examples

Introduction

The definition of “monolithic” is not clear, and it might be more accurate to characterise these wastes as “stabilised” and/or “solidified”. However, for the purpose of this discussion, the word “monolithic” will be used.

A large portion of the waste being landfilled in Sweden appears to be “monolithic” rather than granular. Another large part of the waste does not seem to fall in any of the two categories, granular or “monolithic” (some examples: isolation material, demolition wastes after fires, rubber from conveyer belts, mattresses, hardened glue). This appendix presents some “monolithic” landfills in Sweden, and some landfills that are similar to monolithic landfills.

The most commonly encountered type of “monolithic” landfill in Sweden seems to be landfills with stabilised APC residues (fly ash or fly ash mixed with acid gas cleaning residue) from MSW incinerators. Two examples regarding soil treatment are also presented, one example where metal contaminated soil was mixed with lime, and one example where residue from soil washing was mixed with cement.

Landfilling of stabilised MSWI APC residues

General

Landfilling of stabilised MSWI APC residues as monolithic waste is generally practised in Sweden. The fly ash is mixed with sludge water or other water and the slurry is landfilled (through Bamberger stabilisation of fly ash or cement stabilisation, (Wänstedt, 2001)). The slurry is either driven to the landfill by truck (or deposited directly in a slope, example: Boden Energi AB), or pumped directly to the landfill. The slurry is left to harden in the landfill. At some landfills the slurry is compacted after being deposited (example Högbytorp), while the only compaction occurring at others is that resulting from the necessary driving on the surface (example Boden). Landfilling of stabilised MSWI APC residues has been confirmed for Sofielund, Högbytorp, Boden Energi and Renova, but the survey has not been complete and more landfills with stabilised MSWI APC residues are likely to exist in Sweden. Below, three of these landfills are described in more detail.

The Högdalen incineration plant/ Sofielund landfill

In the 1990s solidified fly APC residues from the MSW incinerator Högdalenverket was landfilled at Sofielund. A slurry consisting of APC residues (from the semidry APC process), special cement, and water was transported in tank trucks (designed to carry sludge) and poured into cells. The cells were divided by 2,5 m high casting walls in order to minimise new surfaces that could leach. The slurry started to harden after a

few days, and the slurry looked homogenous and concrete-like after solidification (Sundberg och Tuutti, 1994).

The Sofielund landfill has been studied in detail through full scale field observations, a pilot scale landfill and laboratory tests. The study included measurement of metal concentrations in water from the full scale landfill (a combination of surface run-off and percolating water); in surface runoff at the pilot scale landfill; and in percolating water at the pilot scale landfill. In the laboratory the following leaching tests were done: batch leaching tests on pure APC residue and on crushed solidified mixture, percolation tests on crushed solidified mixture, uniform leaching procedure and drip-tests on solidified ash.

The permeability and stability of the material was also studied, both in full scale, pilot scale and in the laboratory. A water balance over the landfill has also been determined. Temperature and movement have been measured in situ. Drilling cores have been taken, and the permeability and strength have been tested (Sundberg och Tuutti, 1994).

Landfilling of stabilised APC residue has continued after 1994, and more tests have been done. Data from these tests may become available through Karl-Evert Bergström (Bergström, 2004-01-28).

The Högbypörp landfill site

At Högbypörp, approximately 50 000 ton of ashes from coal burning and APC residues from MSW incinerators are landfilled in specific landfill cells. The MSWI APC residue is left for carbonisation for 12 months, and is then mixed with the coal ash. The coal fly ash has been treated with lime and the mixture will therefore react with the APC residues and harden. Information about this site was provided by Kil (2003-11-21).

At present, the landfill, which has a bottom liner, falls into the non-hazardous landfill category. At the landfill, the mixed concrete-like slurry is placed in of 40-50 cm layers at the landfill. Each layer is thoroughly packed. At the landfill surface, a pattern of small cracks is formed. However, according to information from the landfill manager, these cracks appear only at the surface, and are not penetrating to larger depths.

An independent institute the "Cement and Concrete Institute" (CBI) perform continuous quality testing of the mix of ashes. In the quality control process, the permeability and compaction strength are tested. The waste material must meet a permeability standard of 10^{-9} m/s and the compaction strength varies between 20 and 50 MPa, in the tests.

For the present permit for the landfill practice, the waste material was tested using a surface-leaching test.

Bodens Energi AB

Bodens Energi incineration plant stabilises fly ash by mixing with sludge/sludge water in a Fergmet mixer (concrete mixer). The moisture content prior to landfilling is ca 13 %. The mixture is transported by truck to the landfilling site. The annual amount of ash/sludge mixture is around

800 tonnes TS/year. The fuel for the incinerator is sorted waste, waste wood, and some (10%) bio-fuel. Information about this site has been provided by Lundberg (2003-12-02).

The landfill consists of a dug-out pit with rubber lining at the bottom. When the landfill is filled up, it will be covered by rubber lining as well. It is classified as a hazardous waste landfill. Trucks take the ash/sludge mixture to the edge of the landfill pit and empty there. The landfill is filled up from the edge and onwards. The mixture is not packed, but hardens in the landfill and trucks drive all over the mixture after hardening. It is reported that no cracks are visible at the surface.

Rainwater forms surface runoff and infiltrate into the ash. Since the bottom liner prevents drainage, there is usually water standing in the landfill pit. Leaching water quality is controlled in the control program for the landfill. According to the information received, no contaminants have been found. The ash/sludge mixture is deposited in the water. During winter (several months per year), the mixture is deposited on top of the ice that forms in the pit.

The properties of the ash/sludge mixture are not checked regularly.

Landfilling of stabilised soil/soil-like materials

Landfilling of solidified soil

In Mönsterås municipality, metal contaminated soil has been placed in a landfill for hazardous waste (bottom isolation and top cover with <5 l/m²/year leaching). The soil was mixed with limestone (first stage of the project) in order to increase the buffering capacity. Later, the soil was mixed with CaO instead of limestone, due to high precipitation (the wetness of the soil hindered packing when limestone was used). Information on this landfill has been provided by Mönsterås Kommun (2001).

The soil/lime mixture is placed in layers on the landfill. Each layer is packed separately. The resulting landfill contains 112 000 tonnes of soil and 4 000 tonnes of lime (ca 50 000 m³). Construction of the landfill used a further 42 000 m³. During the construction of the landfill, 4000 m³ water leached out (total 60 weeks). After the construction has been completed, only a few m³/year are expected to leach out from the landfill.

Landfill of solidified soil washing residue

When contaminated soil is treated by washing, a residue of fine-grained material in which the contaminants are concentrated is produced. At one landfill in Sweden, such residues have been treated with cement and landfilled. The mixing with cement is done at the landfill. The resulting slurry was placed in horizontal layers on the landfill so that a mountain of solidified residue was created. According to observations, there were some

cracks in the surface of the landfill, but most precipitation becomes surface runoff.

Based on the EWAC classification, the landfill is classified as a landfill for hazardous waste. No leaching tests have been performed on the solidified residue. The landfill in question has been filled up and will be covered soon. Information on this landfill was provided by Rönnols (2003).

Monolithic-like example – foundry sand with bentonite

About 100,000 to 125,000 tonnes of foundry sand with bentonite need to be disposed of each year from the Swedish foundry industry (Projekt AIS 32, 2003). In Skövde, the sand is used to fill an industrial site. The Skövde foundry sand has a very low permeability (10^{-10} m/s, which is lower than the permeability that is required for the isolating cover of non-hazardous waste: 10^{-9} m/s, and the permeability where the percolation leaching test for testing of granular waste is deemed possible: 10^{-8} m/s).

Due to the low permeability the foundry sand may be looked upon as monolithic. The material has been tested using batch leaching tests for granular waste (EN 12457). A discussion on the interpretation of the tests, and whether convection-based models for metal transport are applicable is currently in progress.

The landfill consists of (from the top down) a layer of foundry sand followed by 0.2 m slag (granular, with high permeability) followed by 0.6-0.8 m of foundry sand. The site is intended for reuse as an industrial building site. Information on this landfill was provided by Rogbäck (2003).

References for Appendix 4

- | | |
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Appendix 5

Release modelling of monolithic landfill scenarios

1. Introduction

1.1 Objectives

The objectives of this part of the project are to:

- Suggest a conceptual model of a monolithic waste landfill,
- Suggest a simple mathematical model for leachate production and contaminant release based on the conceptual model.
- Carry out some preliminary modelling for a selected number of scenarios defined in the conceptual model

The following questions are addressed:

What is controlling the aqueous phase concentration in the landfill? For which substances, and for which flow regime scenarios is it reasonable to assume that the concentration is solubility controlled? Under which conditions will kinetics (chemical and physical) have to be taken into account?

1.2 Conceptual model

The conceptual model represents a qualitative description of the system and may include initial and boundary conditions, state parameters (such as physical and chemical properties of the medium, pH, redox-potential, etc), processes (water flow, dispersion, diffusion, precipitation/dissolution, sorption, degradation, etc), their interactions and influence on events and impacts at the system boundary. Based on the conceptual model, mathematical models may be developed for the purpose of testing hypotheses and assumptions made in the conceptual model and, ultimately, to predict emissions.

In order to enable a quantitative description of a system, simplifications have to be made. The complexity of the system, as described in the conceptual model, must be reduced to something that can be handled mathematically. Still it must be complex enough to capture the essence of the system characteristics. Obviously there is not one unique way of developing a conceptual model based on the available basic information. Nor is there one unique way of developing a mathematical model based on the conceptual model. Developing a conceptual model is therefore often an iterative procedure.

1.3 Uncertainty

The uncertainty in predictive modelling may be attributed to uncertainty in the design of the conceptual model, uncertainty and variability in the input parameters and limitations in the mathematical models and numerical algorithms. The uncertainty in the input parameters is due to limited data and knowledge, while variability is caused by short or long range variations in time and space. Parameter variations in time may be a result of slow changes of the physical and chemical properties (aging) and variations of the boundary conditions such as rainfall, temperature, mechanical impact etc. The spatial variations are due to the heterogeneity (in different spatial scales) of the material.

The relative effect of these uncertainties on the performance of the mathematical model depends on the studied system, but the uncertainty in the conceptual model dominates in many cases. It is, to some extent, possible to quantify the effects of parameter uncertainty and variability by sensitivity and uncertainty analysis. Also, uncertainty associated with the mathematical model may be assessed. However, uncertainty in the conceptual model is more difficult to assess.

2 Conceptual model monolithic landfill

2.1 Geometry, crucial system parameters and boundary conditions

The aim here is to give a first qualitative description of the processes, emission scenarios and boundary conditions that are crucial for assessing the leaching of contaminants from a monolithic landfill.

Based on the former work within the TAC modelling group the model landfill is assumed to have a 200 m * 200 m * 20 m geometry with a density of 1.8 t/m³.

In some landfills for monolithic waste the stabilised waste is placed in sections and layers, separated by geotextile or granular material in order to create preferential flow paths, which will take care of any infiltrating water and rapidly channel it downwards. In other cases the stabilised waste is just put in place, as solid monoliths or as slurry which will harden in situ. In both cases it seems reasonable to conceptualize the landfill as a two-domain medium, composed of a matrix and preferential flow paths or channels. Homogeneous blocks of stabilised waste constitute the matrix domain. In the matrix domain, capillary forces govern the water movement and the dominant transport process is diffusion. A network of flow paths, which may either be designed channels or fractures that have evolved randomly, define the other domain. In the fracture domain both gravity and capillary forces govern the water flow. The flow may take place either as a saturated flow or as a film flow along the boundaries of the channels/fractures. In a Swedish study (Sundberg and Tuutti, 1994)

the hydraulic conductivity in landfilled cement stabilised fly was about two times higher in the horizontal than in the vertical direction. The fractures were mostly occurring in the horizontal space between different layers of cement ash.

Figure 1 illustrates the conceptual model of a monolithic landfill. The height of the landfill (h) is a function of time until the landfills is filled and the final cover is put in place. The cover plays an important role for the leachate production (Q_L) as it governs the split of precipitation (P) into evaporation (E), surface runoff (Q_{sur}), storage (S) and infiltration (I).

The conceptual model must be very simple for practical reasons. Not all of the parameters listed in table 1 will be taken into account in the model and for the parameters that are included in the model general values have to be chosen. For parameters that are mutually dependent one parameter may be expressed as a function of another. In this way, the number of model parameters may be reduced. To give an example; fractures develop to relieve the tensions induced by shrinking, swelling or external load. The total “fracture volume” may either be distributed on a few fractures with large apertures or on many fractures with small apertures. It may therefore be reasonable to assume that the fracture aperture is proportional to the fracture spacing.

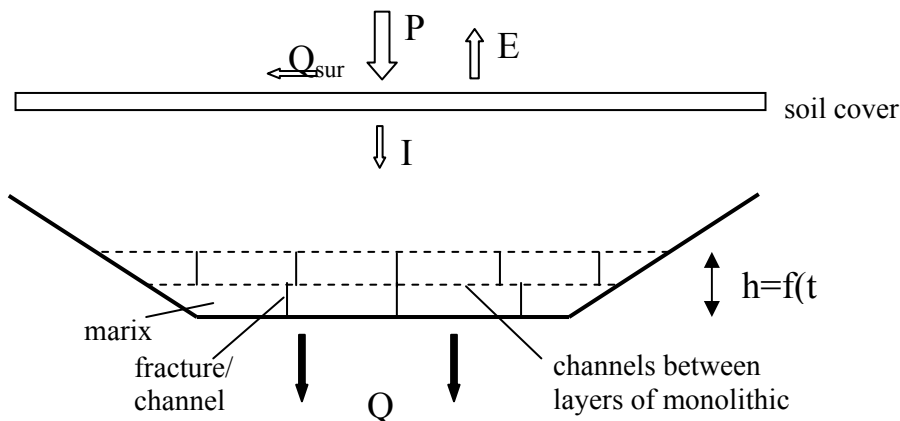


Figure 1
Schematic illustration of a monolithic landfill.

The crucial physical parameters for the two-domain system are summarized in table 1:

Table 1 Important physical parameters.

Fractures/channels	Matrix
Spacing	Density
Aparture	Porosity
Cross-sectional geometry	Effective diffusivity coefficient
Length	Retention (pF-) curve for matrix
Connectivity	Surface area of the boundary between the fracture and matrix
Orientation	Hydraulic conductivity
Hydraulic conductivity	

2.2 Leaching scenarios

The leaching process can be controlled by:

- Availability
- Solubility
- Kinetics in chemical and physical processes

Two simplified leaching scenarios can be distinguished which will cover the influence of these three controlling mechanisms. These two scenarios are illustrated schematically in figure 2.

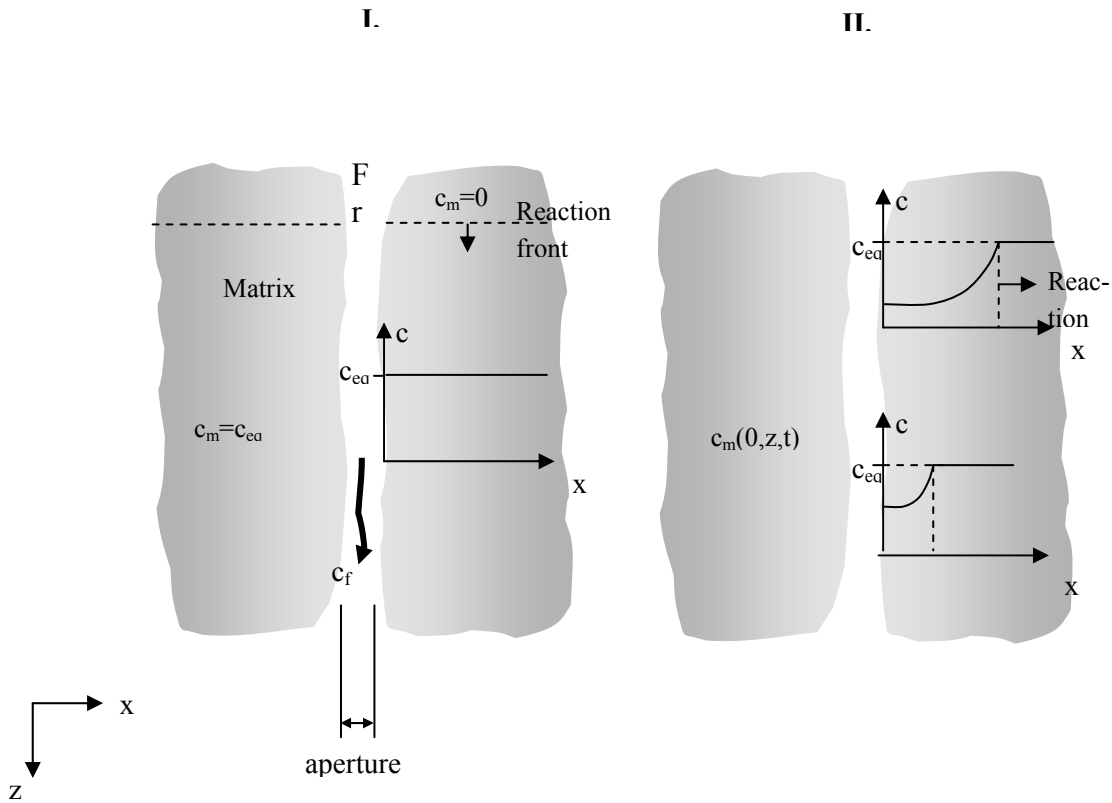


Figure 2
Two simplified leaching scenarios

In **scenario I** the velocity in the fractures is slow compared to the kinetics in the chemical reactions and the diffusion process. This makes the local equilibrium assumption (LEA) valid. The aqueous phase and the solid phase are in equilibrium and the concentration in the matrix pore water (c_m) is equal to the concentration in the fracture domain (c_f). Depending on the substance, the equilibrium concentration may either be defined by availability or solubility. As the controlling minerals are exhausted a reaction front will develop and travel downwards.

In **scenario II** the velocity in the fractures are fast compared to the kinetics in the chemical reactions and the diffusion process. This creates a non-equilibrium situation and diffusion and chemical reaction rates become significant retardation mechanisms. In this case the aqueous phase concentration will be a function of x , z and t . Reaction fronts will evolve from the fracture and travel horizontally into the matrix.

To conclude, these two scenarios illustrate how the leachate concentration at the lower boundary of the landfill is dependent on the characteristics of the flow regime in combination with availability, solubility, diffusion, and reaction kinetics.

2.3 Generation of leachate: Solubility/equilibrium control or kinetic control?

Two simplified leaching scenarios (described in section 7.1 in the main report) and the influence of the flow velocity is here explored roughly with a simple model. The following question is addressed: Under which conditions is it reasonable to assume that the concentration of a given component in the leachate at the lower boundary of the landfill is solubility controlled and when will kinetics (chemical and physical) have to be taken into account?

It is here acknowledged that the release rate, which is observed in tank leaching tests such as NEN 7345, is governed by the synergistic effect of diffusion, reaction kinetics and solubility control in the monolith interior. The transport process in a landfill of monolithic waste is assumed to be dominated by diffusion from the matrix to the fracture domain where the dissolved species are rapidly transported out from the system by advection. As a first approach, the release rate from the matrix domain into the fracture domain is here assumed to take place as a zero order process (constant release rate) constrained by a solubility limit in the fracture domain for the substance in question. The convection-dispersion equation without the dispersion term is here applied to describe the transport process in the fracture domain under steady state conditions:

$$\theta \frac{\partial C}{\partial t} + \frac{q}{1000} \frac{\partial C}{\partial z} + \rho \cdot \frac{\partial S}{\partial t} = 0$$

where

$$q = \frac{I}{365}$$

and

$$\frac{\partial S}{\partial t} = -k\phi \left(1 - \frac{C}{C_{eq}}\right)$$

C is the concentration (mg/l) in the fracture domain, C_{eq} is the solubility limit (mg/l), θ is the water content in the fractures per volume of the fractured media, q is the infiltration rate of rain water (mm/day), I is the annual infiltration (mm), ρ is the density (kg/l). S is the available mass density of a certain substance (mg/kg), k is the rate coefficient (mg/m²/day) for the zero order release process and ϕ is the geometric surface (m²/kg) of the matrix that is exposed to moving water, that is the boundary between the fracture and matrix domain.

The initial and boundary conditions are:

$$C(z,0) = 0, \quad 0 \leq z \leq Z$$

$$C(0,t) = 0, \quad 0 \leq t$$

where Z is the total height of the landfill (m).

Combining the equations and solving with the method of characteristics yields the following solution for the concentration at the lower boundary of the landfill as a function of infiltration rate:

$$C = C_{eq} \left(1 - e^{-\frac{\rho k \phi T}{\theta C_{eq}}}\right) = \left[T = \frac{Z \cdot 1000}{v} = \frac{Z \theta \cdot 1000}{q} \right] = C_{eq} \left(1 - e^{-\frac{\rho k \phi Z \cdot 1000}{q C_{eq}}}\right) = C_{eq} \left(1 - e^{-\frac{kAZ}{q C_{eq}}}\right)$$

where T is the time (days) it takes for infiltrating water to travel from the surface to the lower boundary of the landfill, $A = \rho \phi \cdot 1000$ is the geometric surface area per unit volume (m^2/m^3) of monolithic waste that is exposed to the water in the fractures. The parameter A depends on the geometry and the scale, for example $A = n/a$ for a cube, where n is the number of sides which are exposed to water and a is the side length (m).

Let us assume a landfill, 20 m high and composed of monolithic waste cubes, with the side length of 0.5m, separated by a channel/fracture network. The parameter A becomes equal to 12 (m^2/m^3).

The concentration at the lower boundary of the landfill, expressed as C/C_{eq} , is here calculated for three different values of the rate coefficient, k , and is shown in figure 3 as a function of the annual infiltration. In this example the solubility limit, C_{eq} , is set to 1 mg/l.

This simple calculation clearly demonstrates the influence of infiltration rate, solubility control and release rate from the monolithic matrix. For example (given the assumptions and input data that was used in this case), figure 3 shows that release rates of about $1 \cdot 10^{-3} \text{ mg}/\text{m}^2\text{day}$ will result in concentrations governed by solubility for all annual infiltration rates lower than about 20 mm.

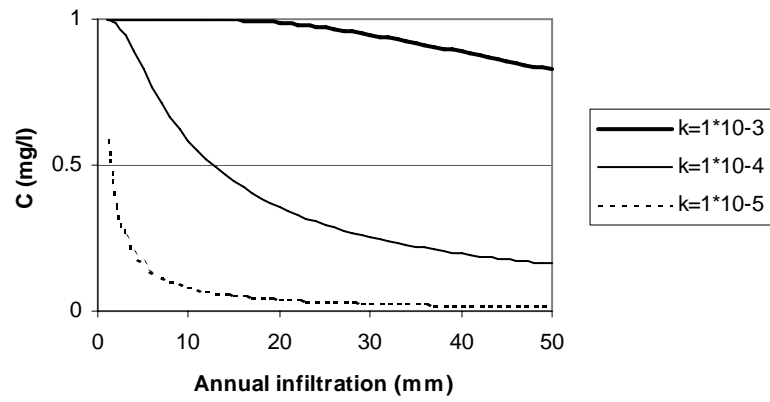


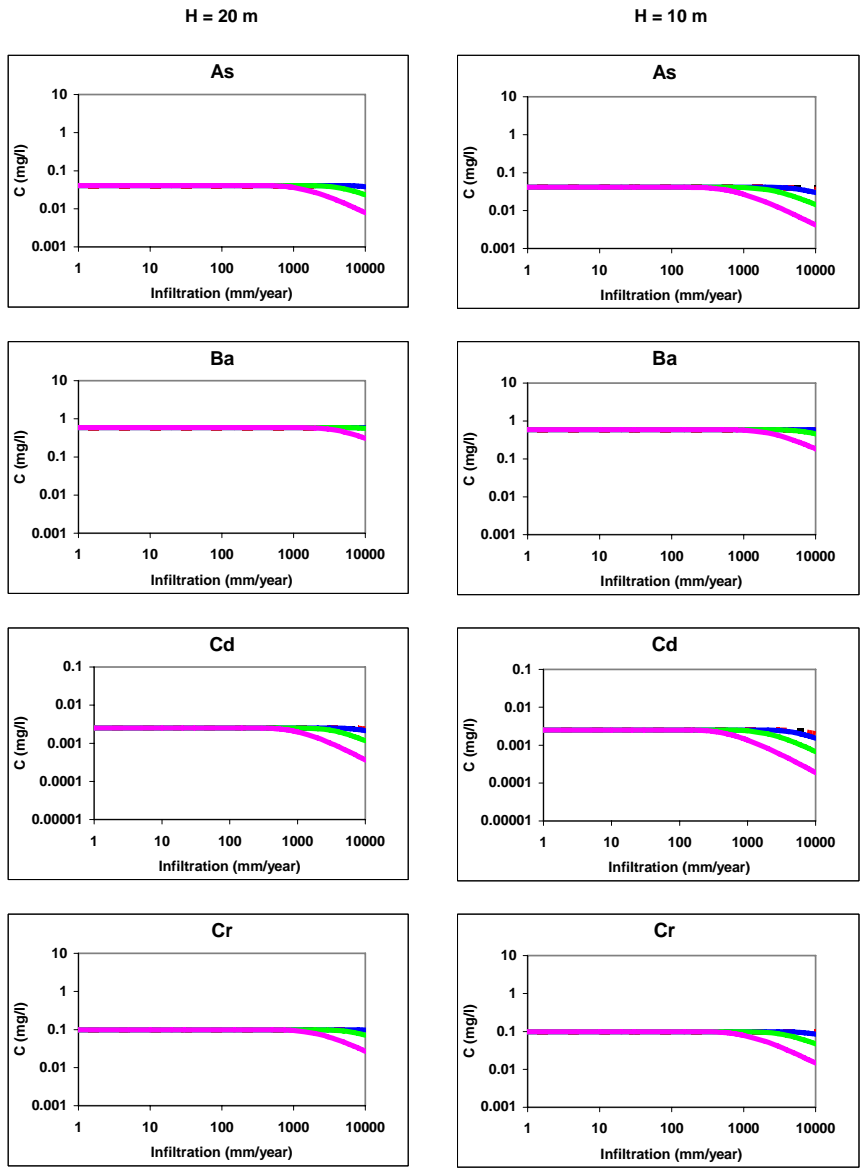
Figure 3
 The concentration at the lower boundary of a monolithic landfill as a function of infiltration rate (mm/yr) for different values of k (mg/m²/dy).

Reference

Sundberg J., K. Tuutti, Solidifiering av rökgasreningssprodukt från Högdalenverket, Slutrapport, Terratema, Linköping, 1994

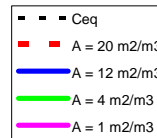
Appendix 6

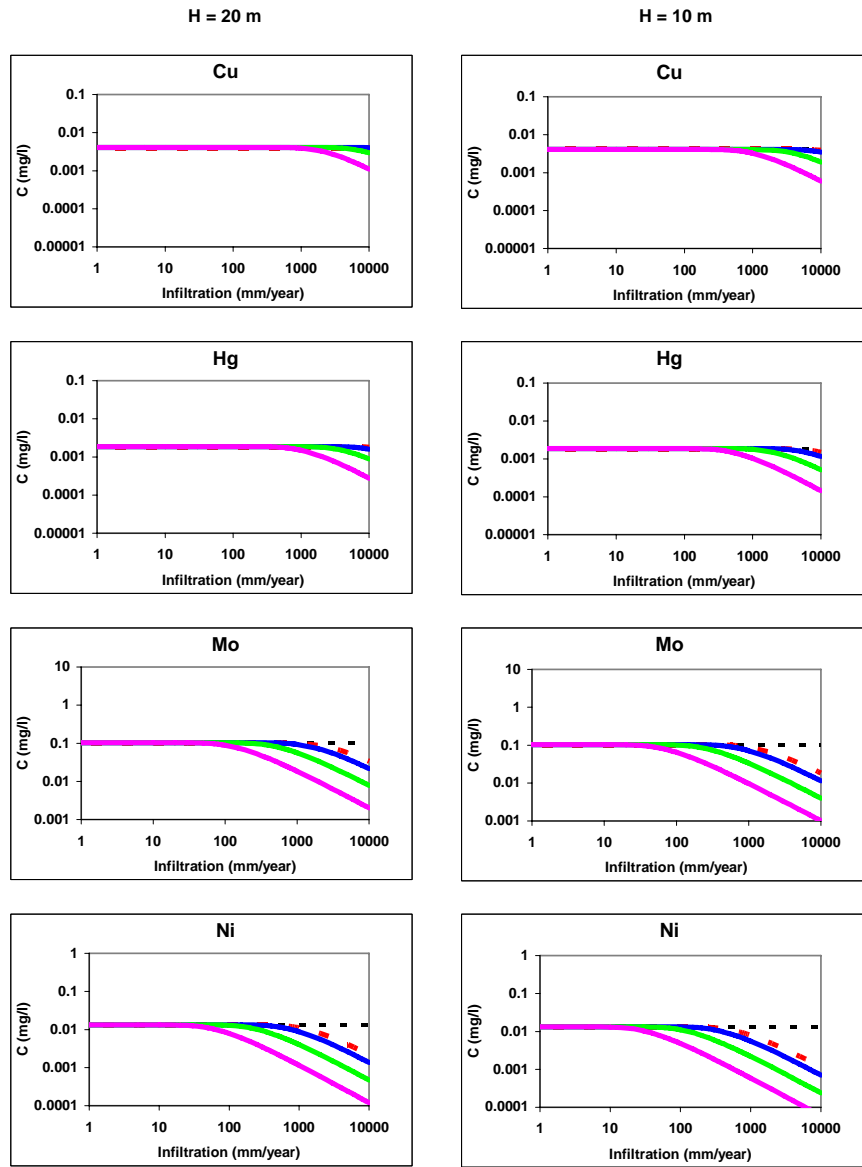
Test of the equilibrium assumption for three stabilised waste materials in landfill scenarios



Test of equilibrium assumption for **StabW**

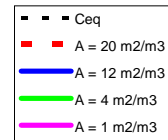
C_{eq} is taken from table 7.2
 k is taken from table 7.3

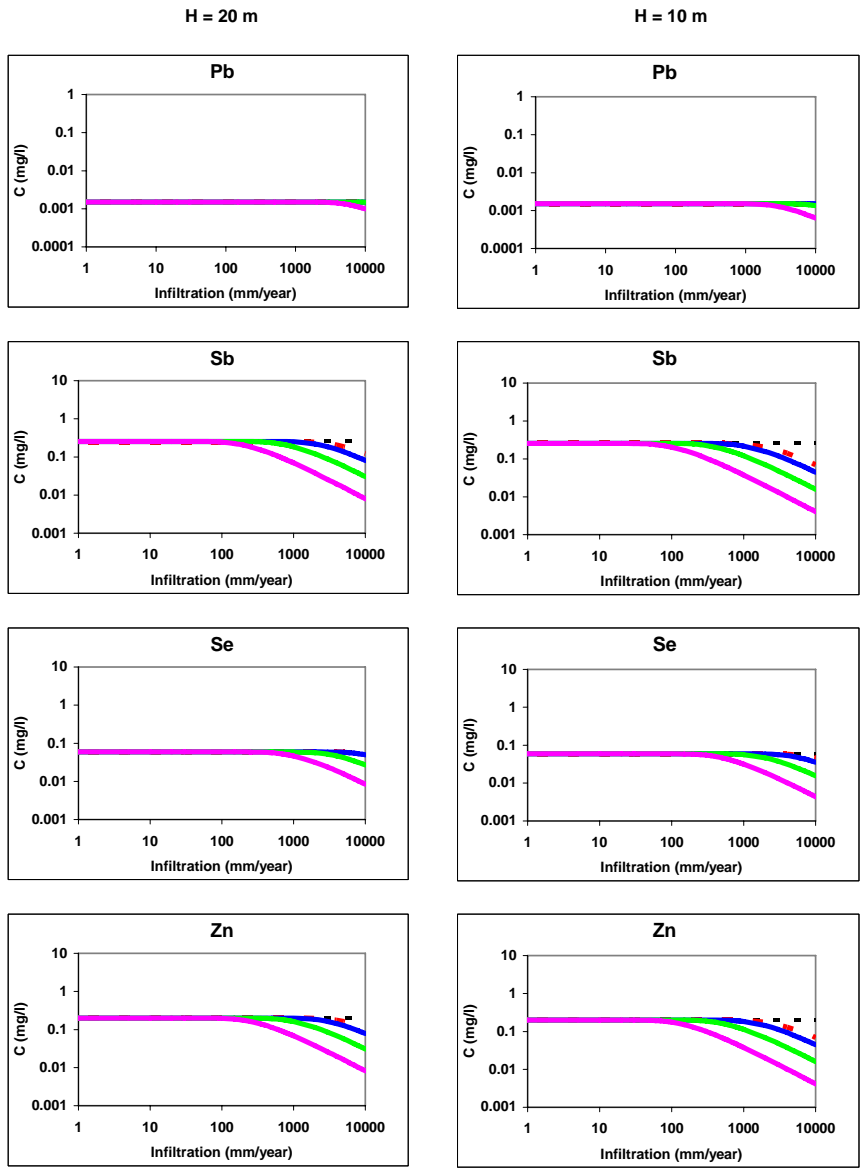




Test of equilibrium assumption for **StabW**

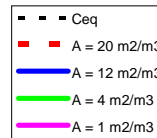
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k is taken from table 7.3

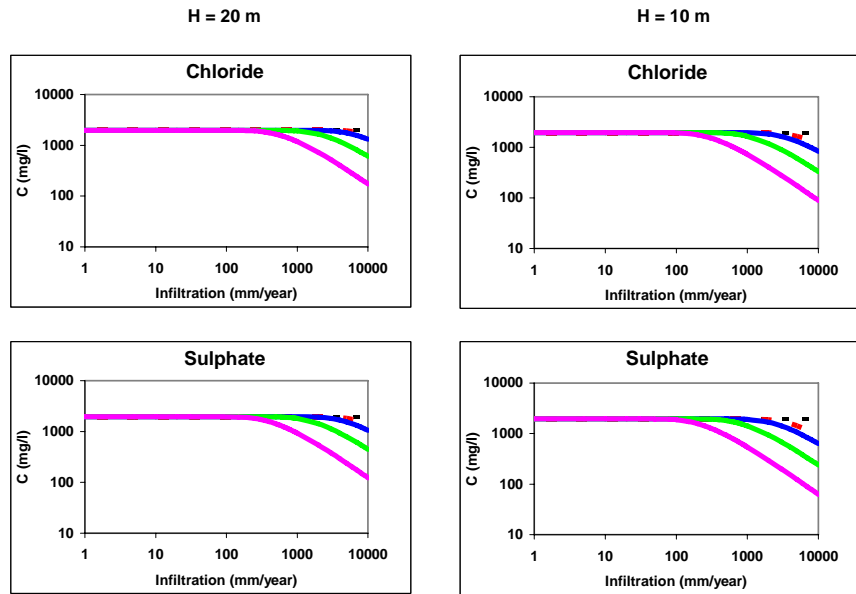




Test of equilibrium assumption for **StabW**

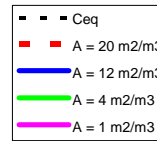
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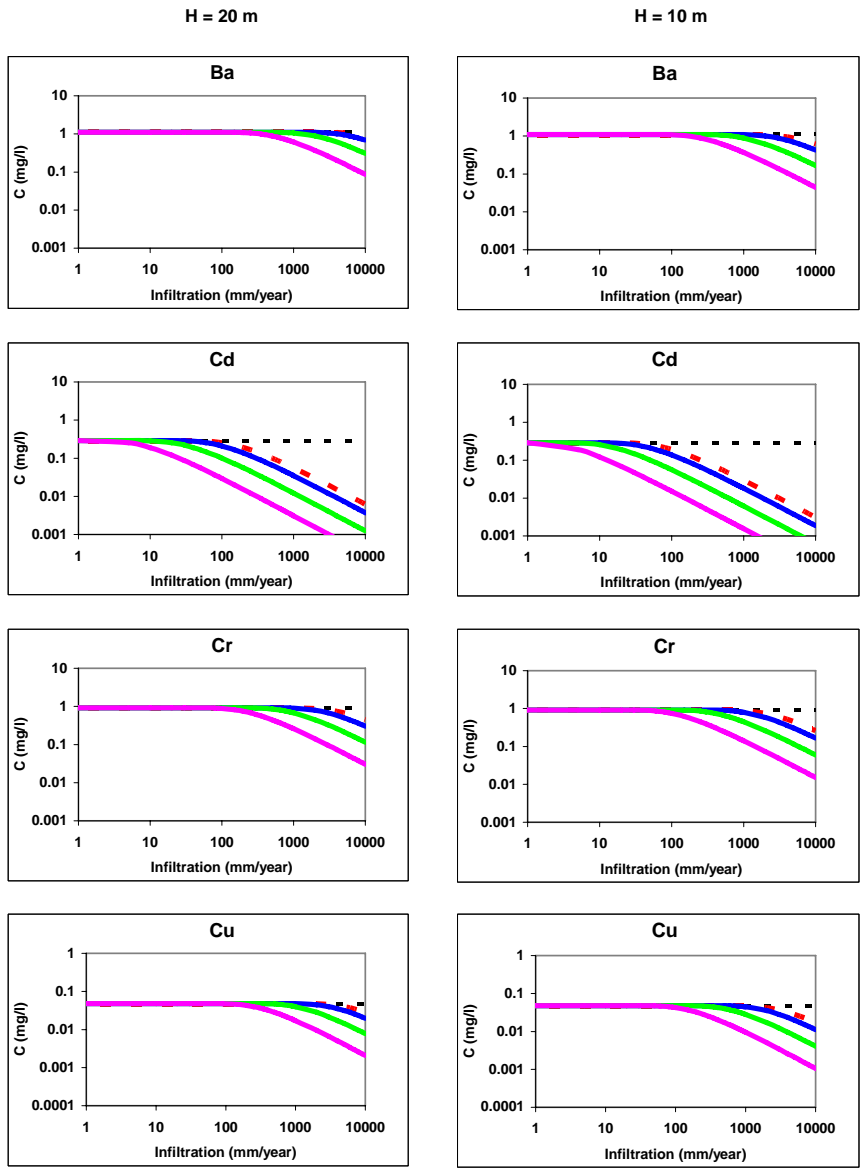




Test of equilibrium assumption for **StabW**

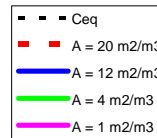
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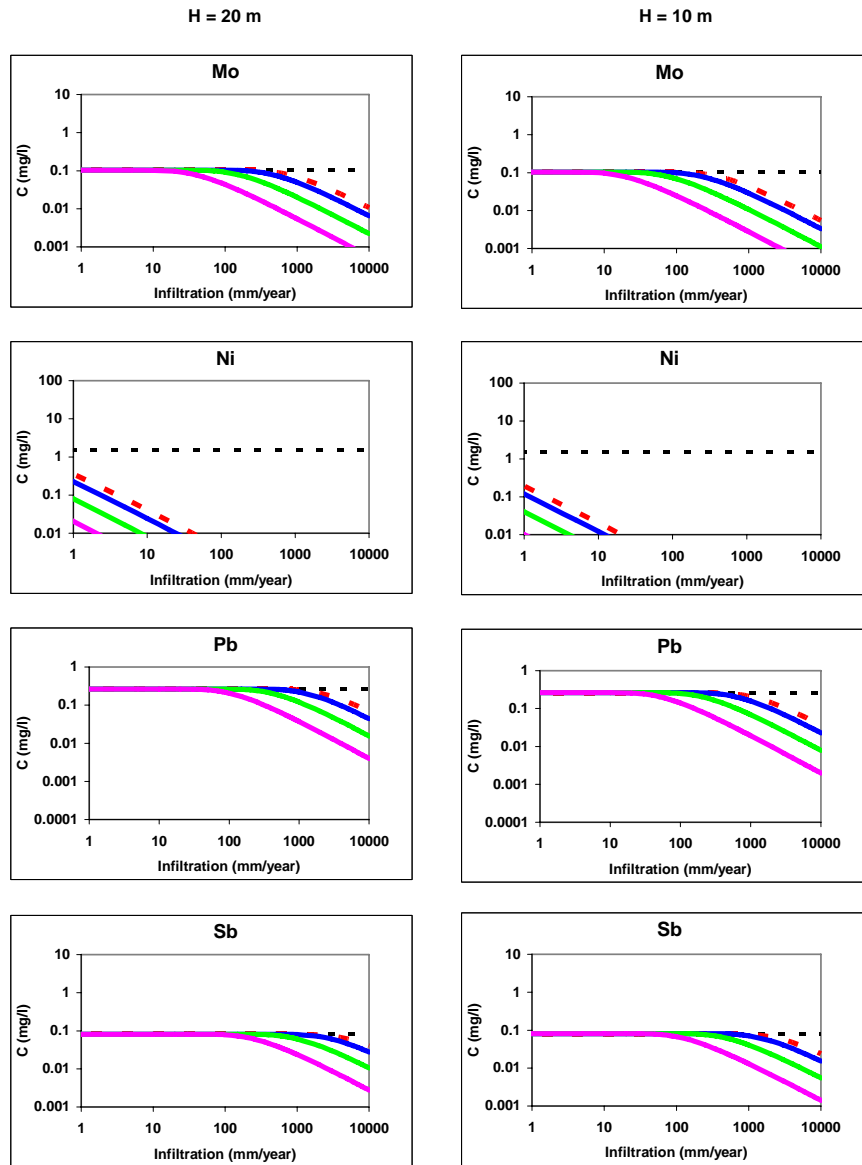




Test of equilibrium assumption for **BCR-1**

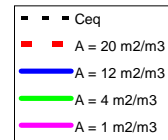
C_{eq} is taken from table 7.2
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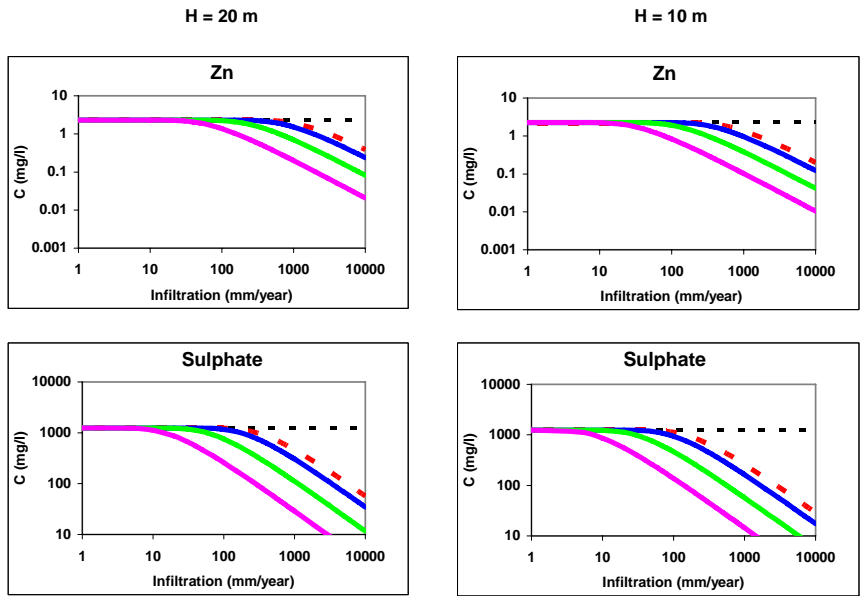




Test of equilibrium assumption for BCR-1

C_{eq} is taken from table 7.2
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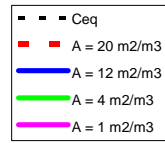


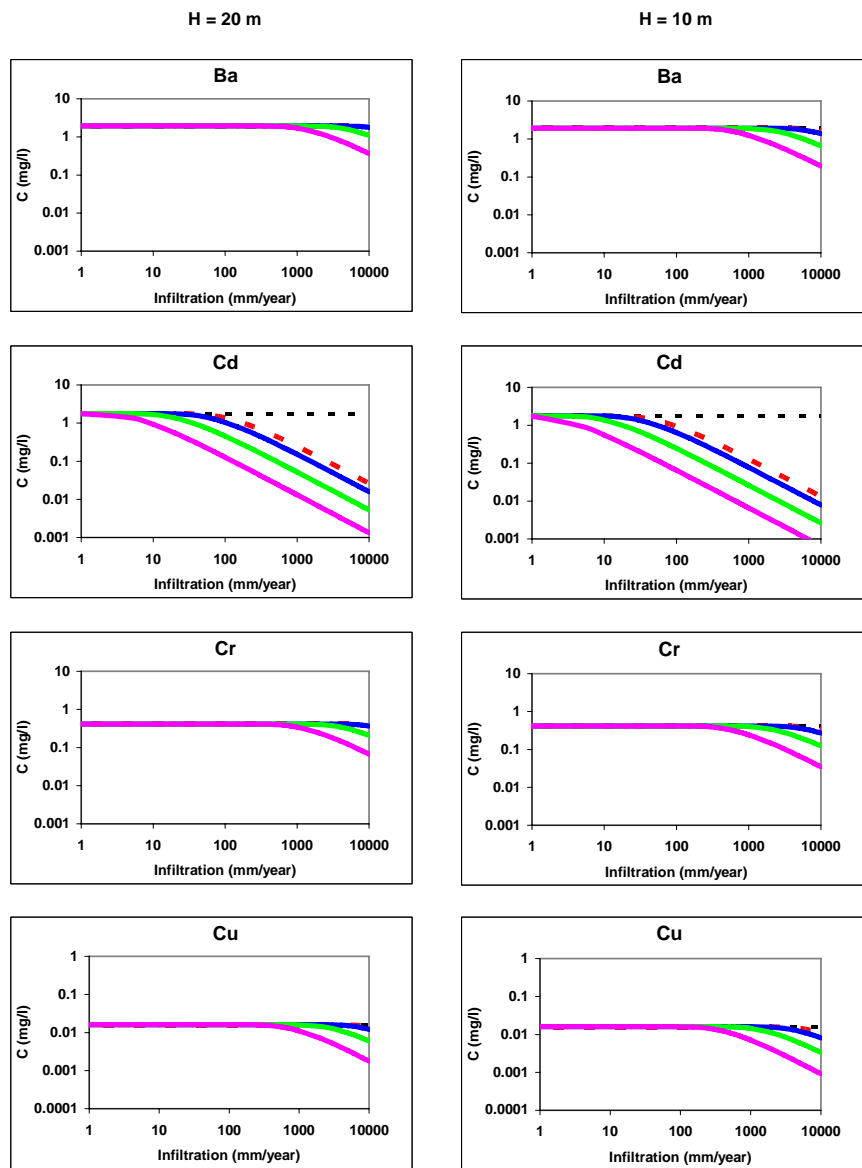


Test of equilibrium assumption for **BCR-1**

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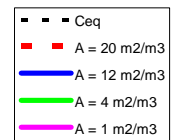
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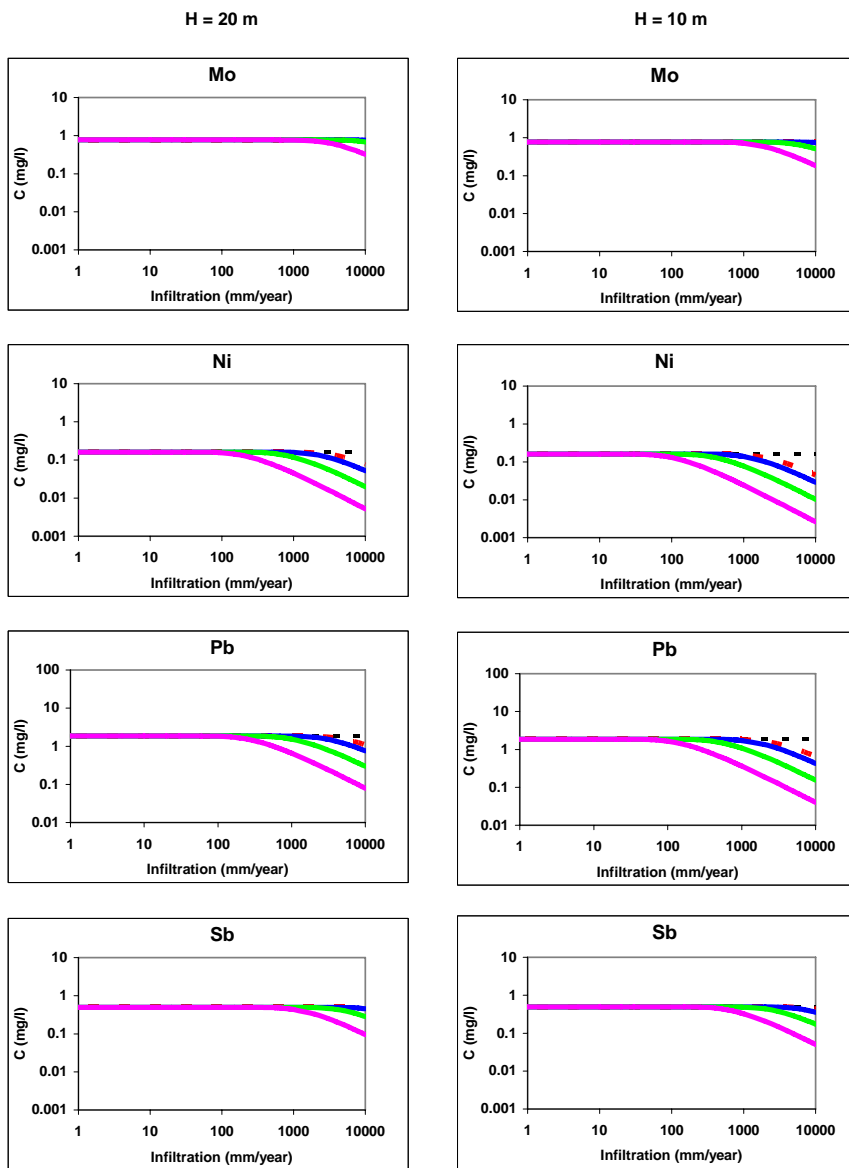




Test of equilibrium assumption for BCR-2

Ceq is taken from table 7.2
k is taken from table 7.3





Test of equilibrium assumption for **BCR-2**

C_{eq} is taken from table 7.2
 k is taken from table 7.3

